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SARR, D.A.

ECOSYSTEM RECOVERY FROM LIVESTOCK GRAZING IN THE SOUTHERN SIERRA NEVADA

FS Contact: Dr. David Cole

CoOp Contact: Dr. Thomas Dudley

FINAL REPORT FOR RESEARCH AGREEMENT
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UNIVERSITY OF CALIFORNIA
FS Contact: Dr. David Cole
CoOp Contact: Dr. Thomas Dudley

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Aldo Leopold Wilderness Research Institute
Box 8089, 790 Beckwith Ave.
Missoula, Montana 59807

Attn.: David Cole

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Ecosystem Recovery from Livestock Grazing in the Southern Sierra Nevada

Authors:

Daniel Sarr,, Roland A. Knapp, Jonathan Owens*, Teri Balser*, and Tom Dudley**

Marine Science Institute, University of California, Santa Barbara, CA 93106

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* Civil Engineering/Environmental Science, Policy & Management, University of California, Berkeley, 94720

** Integrative Biology, University of California, Berkeley 94720

INTRODUCTION

Of the eleven co-terminous states of the western U.S., 90% of the public lands (approx. 285 million acres), or 48% of the total area, are grazed by livestock (Armour et al. 1991). Recent range surveys on Bureau of Land Management (BLM) grazing allotments have described a majority (58%) as being in fair to poor condition (Armour et al. 1991), and a National Research Council analysis of rangeland health concluded that not only is the extent of ecological damage underestimated by most range assessments, but that degradation is continuing (NRC 1994). Increasingly, managers, ranchers, and conservationists have targeted riparian ecosytems as indicators of rangeland health because they are among the most sensitive of western ecosystems to livestock impacts, and the most responsive to modified management (Kattleman and Embury 1996, Menke et al. 1996). Although riparian habitats are usually small in area, they are crucial to the maintenance of biodiversity, especially in arid regions (Naiman 1993). Livestock strongly prefer riparian zones, where their impacts are intensified by moist soil and fluvial disturbance (Mosconi and Hutto 1982, Szaro 1989, Chaney et al. 1990, Williamson et al. 1992, Elmore and Kauffman 1994, and others), resulting in adverse effects on biological communities, especially coldwater fisheries (Behnke and Zarn 1976, Naiman 1993). Small headwater streams, which usually originate on public lands, are particularly vulnerable to livestock induced erosion (Williamson et al. 1992, Quinn et al 1992), and the effects of such damage can accumulate down watersheds (Burns 1991).

In montane meadows of the western United States, early grazing practices have led to destabilization and downcutting of stream channels (i.e. gully formation), lowered water tables, and vegetation change from herbaceous sod (particularly grass/sedge meadow vegetation) to xeric shrub stands generally dominated by sagebrush (*Artemisia* sp.) (Kauffman and Krueger 1984, Wood 1975, Odion et al. 1988, Chaney et al. 1990, Allen-Diaz 1991). These altered ecosystems are more prone to continued erosion as floodflows are confined within incised channels rather than dissipating energy over a natural, wide floodplains (Rosgen 1994). Incised stream channels also lack the structural features necessary to retain suspended sediment (Dieter 1990, Lusby 1970), allowing increased sediment flux and instability within aquatic habitats.

As in the rest of the western U.S., most riparian meadows in the Sierra Nevada have been grazed almost continuously by domestic livestock for over 100 years, and long-term impacts, such as extensive gully erosion, altered stream geomorphology, lowered water tables, and loss of wet meadow habitat, have been documented throughout the range (Wood 1975, Hagberg 1994, Dudley and Embury 1995). What remains poorly understood is how to promote recovery of these systems so that they are able to sustain native assemblages of plants and animals, while retaining the resiliency to withstand both natural and anthropogenic stresses, including livestock production. Public land managers have begun implementing and studying riparian zone fencing as a way to restore the health of these ecosystems (Platts 1991, Kondolf 1993). This study was designed to document processes and evaluate the effectiveness of ecosystem recovery in sub-alpine meadows of the southern Sierra Nevada after short-term livestock exclusion.

Grazing history and meadow erosion in the southern Sierra Nevada

Montane meadow ecosystems are vital elements of the Sierra Nevada landscape, providing wildlife habitat, water and sediment retention, livestock forage, and varied aesthetic and recreational opportunities (Ratliff 1985). Most are fluvially formed features supporting characteristic plant species that tolerate high water tables, at least during a significant part of the year (Ratliff 1985). Meadows of many sizes occur throughout the middle and high elevations of the range, and large meadows are particularly important in basins not deeply scoured by Pleistocene glaciation. The high productivity within these habitats, and availability of green forage in summer, has made them preferred summer range for domestic livestock (Ratliff 1985).

In most meadows of the southern Sierra Nevada, domestic livestock grazing has been the dominant land use since the mid 1800's (Ratliff 1985). Sheep production was initially the primary usage in these meadows, and sheep were grazed in virtually every accessible meadow in the southern Sierra Nevada. Although quantitative data on the effects of early livestock grazing are unavailable, anecdotes from travelers suggest that impacts were widespread and severe. The following quote from the mountaineer Clarence King (1902) is particularly telling:

"The Kern Plateau, so green and lovely in my former visit in 1864, was now a gray sea of rolling granite ridges, darkened at intervals by forest, but no longer velveted with meadows and upland grasses. The indefatigable shepherds have camped everywhere, leaving hardly a spear of grass behind them."

By 1900 cattle had largely replaced sheep grazing on mid and high elevation lands (DeBenedetti and Parsons 1979, Hagberg 1994), and grazing intensity has generally decreased over this century (Ratliff 1985, Sarr 1995). On the Kern Plateau, the focal area of this study, cattle and sheep were driven up into the area as soon as the snow melted, when meadows were saturated and most vulnerable to erosion (Ratliff 1985). It is believed that much of the severe gully erosion now evident in the Golden Trout Wilderness (Inyo National Forest) was initiated during these early years of overgrazing. Local anecdotes compiled by Forest Service land managers suggest that much of the dramatic erosion in the area occurred between the 1870s and the 1940s. According to testimony of a local rancher, upper Mulkey Creek, now in a gully up to 5 meters deep, flowed on the surface and meandered through moist meadow for its entire length in the 1890's (USDA Forest Service 1961). With inclusion of the Kern Plateau in the Sierra Forest Reserve in 1905, sheep grazing was prohibited and cattle grazing allotments were established. Inyo National Forest officers estimated that several thousand head of cattle per season grazed the Plateau from the late 1800's until the 1920's (USDA Forest Service 1961). Photographs from this period testify to the severity of erosion at its worst (Figure 1).

The Taylor Grazing Act (1934) and continued adjustments in range management on the Kern Plateau led to substantial decreases in actual livestock use after the 1920s (NRC 1994, Sarr 1995). Presently, seasonal limits are 550 cow/calf pairs on the Templeton Allotment (Templeton and Ramshaw Meadows) and 235 cow/calf pairs on the Mulkey (Mulkey Meadows) Allotment. Cattle are allowed on the allotment as soon as the district range conservationist determines the meadows to be sufficiently dry to resist trampling damage (usually from mid June to early July), and are removed when forage utilization or streambank trampling and chiseling standards are reached in the fall (usually late September) (D. Hubbs, USFS-Inyo N.F., personal communication.).

Beginning in the 1930's, work began to combat the gully erosion and restore natural hydrological function in several meadows of the Kern Plateau. The restoration projects involved primarily construction of log check-dams to stop headcut erosion in active gullies, reseeding of denuded areas, and protection from livestock through fencing or removal of livestock. Despite exceptional efforts of individual managers, funding constraints and changes in management priority limited the breadth and continuity of restoration projects (Sarr 1995). As a consequence, most projects failed due to washout or livestock trampling, and gully erosion has continued in most of the watershed. Riparian fencing is now the preferred method for watershed improvement on the Kern Plateau (D. Hubbs USFS personal communication.). It is now believed that natural vegetation recovery within exclosures will help minimize stream and gully erosion and improve riparian and aquatic habitat more efficiently than stream structures, while providing a useful example of undisturbed habitat.

Investigations of the ecological impacts of grazing have often compared continuously grazed areas with areas recently excluded from livestock use (Fleischner 1994). This less than ideal avenue of research (see discussion below) has been necessary because there are presently very few, if any, pristine rangelands remaining in the American West (Fleischner 1994). These exclusion studies have indicated geomorphological alterations following exclusion, including: decreases in stream width and bank angle with

increases in depth and proportion of undercut banks (Marcuson 1977, Van Velson 1979, Platts 1981, Marlowe et al. 1987). Riparian and aquatic flora have shown increases in plant biomass, density, and species diversity in riparian exclosures (Odion et al.1988, Schultz and Leininger. 1990). Some responses of fauna, including increased salmonid biomass and density, have been documented (Fleischner 1994), yet other studies have shown no change (Platts 1991). Very few investigations have documented within a single study the recovery (or lack thereof) for the many ecosystem components (geology, soils, vegetation, fauna) that interact to determine ecosystem function or resistance to future perturbation (Williamson et al. 1992, Quinn et al 1992). This is needed because there are many site specific factors (such as elevation, basin geology, and vegetation type) and processes (precipitation, magnitude and periodicity of floods) influencing the recovery of stream and riparian habitats which are best discussed simultaneously. The lack of rigorous evaluation of ecosystem recovery is a common concern among restoration ecologists (Westman 1991), suggesting that more restoration efforts should include documentation of both the mechanisms and rates of recovery, and whether such recovery is resistant (sense Holling 1966) to future perturbation.. This study aimed at evaluating a range of geological and biological variables to clarify mechanisms and trajectories of recovery in fluvial meadows of the southern Sierra Nevada.

BACKGROUND

Meadow formation and geomorphology

Mountain meadows are biologically and geologically dynamic. Early views that Sierran meadows are transitory Pleistocene relicts (Bakker 1971) have been replaced by a fluvial equilibrium model. Wood (1975) argued that meadows represent an interactive relationship between a catchment, its surface and groundwater hydrology, and its sediment budget which has fluctuated through time in response to regional climatic conditions and sediment availability. His stratigraphic studies in the Yosemite region showed that many present meadow areas experienced afforestation during post glacial warming (~10,200 years B.P.) followed by meadow reestablishment(~1200 years B.P.) due to rising water tables under our present neoglacial climate. Wood (1975) concluded that most Sierran meadows are aggrading, self sustaining entities under the present climate where sediment capture and on site peat formation meet or exceed losses to erosion. The results of Wood (1975) are supported by a recent study of an eroding wet meadow in the northern Sierra Nevada that revealed extensive evidence of recent forest debris (< 150 years old) under wet meadow organic deposits (Bartolome et al. 1990) indicating advances of meadow habitat until relatively recently.

Benedict (1982) proposed that meadows can be classified as geologically stable or unstable. In geologically stable meadows, such as aggraded glacial basins resting on bedrock, perturbations of vegetation will have little lasting effect upon the meadow's structure. In contrast, most meadows of fluvial origin are geologically unstable, but are biologically stabilized by vegetation. and can be severely degraded by vegetation disturbance. In fluvial meadows, stresses that lower vegetative cover or biomass can lead to rapid channel entrenchment and sediment loss (Swanson 1988).

Study Area

To improve our understanding of recovery of degraded meadow ecosystems, we conducted a study of subalpine fluvial meadows of the South Fork Kern River watershed. The upper portion of this watershed was designated as Wilderness within the Golden Trout Wilderness Area in 1977, in part to protect and restore the habitat of the endemic California golden trout (*Oncorhynchus mykiss aguabonita*; a conspecific sub-species, *O. m. whitei*, is endemic to an adjacent watershed). The Golden Trout Wilderness (GTW) contains many deeply incised stream channels, unstable aquatic habitats, and former wet meadow

habitat now occupied by xerophilic species such as sagebrush (*Artemisia* sp.). In an attempt to remediate livestock impacts to several meadows, the Inyo National Forest constructed large exclosures in three meadows between 1983 and 1991 (Figure 2). The attributes of each exclosure are summarized in (Table 1) We used these livestock exclosures to study the process of recovery of several potentially interacting ecosystem components, including channel geomorphology, soils, riparian vegetation, aquatic invertebrates, and fish.

The unglaciated subalpine watersheds of the Kern Plateau (118° 15' N, 36° 22' W) contain one of the most extensive meadow complexes in the Sierra Nevada. In contrast to the dramatic glacial topography of the "High Sierra" to the north, the Kern Plateau is characterized by undulating terrain and open, park like fluvial basins separated by timbered granite ridges.

The research described in this paper was conducted primarily in three large, low-gradient meadows (Mulkey - elevation 2806 m, Ramshaw - elevation 2597 m, and Templeton - elevation 2567 m; Figure 2, Table 2) in the eastern half of the GTW. As is typical for the large meadows of the Kern Plateau, they are classified by Benedict and Major (1982) as Type II physiographic meadows (i.e., occurring only in the Sierra Nevada near the southern limit of historic glaciation).. Meadows are underlain by granitodiorite alluvium and fringed by coarse, sparsely vegetated granitic sands (Albert 1982). Meadow soils, similar to those described by Bennett (1965) and Wood (1975), are derived primarily from granitic fluvial deposits interspersed with peat layers and occasional volcanic ash deposits. Topsoils, where they have developed, are slightly acidic sands, loamy sands, and occasionally sandy loams (Sarr 1995). Open forests of lodgepole pine (*Pinus contorta* ssp. *murrayana*) occur above the alluvial margins and grade into foxtail pine (*Pinus balfouriana*) on surrounding ridges. Forested areas are separated from moist meadows by stands of sagebrush (*Artemisia. cana, A. tridentata,* and *A. rothrockii*) and other xeric species (Ratliff 1985, Odion et al. 1988, Sarr 1995). Streamside zones are typically occupied by a rich mosaic of herbaceous vegetation communities differentiated by gradients in moisture, soil texture, and time since fluvial disturbance (Sarr 1995). Sedges (*Carex* sp.), hydric grasses (*Deschampsia, Alopecuris* sp.), and pioneering forbs (*Gnaphalium, Epilobium* sp.) occupy streamside areas. Perennial herbs (*Erigeron, Gentiana, Rumex* sp.), rushes (*Juncus* sp.), and upland grasses (*Poa, Trisetum, Muhlenbergia* sp.) become important with increasing elevation above the stream channel. Although abundant in steeper drainages nearby, willows (*Salix* sp.) are relatively rare in the study meadows, occuring at middle elevations on the floodplain (Odion et al. 1988, Sarr 1995).

Precipitation is approximately 50 cm per year, with over 90% falling as snow (Benedict 1983, Albert 1982, Odion et al. 1988). Average maximum snowpack depth is 1.4 m with 59 cm water content for Siberian Outpost, another Type II meadow 27 km northwest of our sites and about 300 m higher (Ratliff and Westfall 1992). Snow melt and peak runoff occur from late March to early June, with large variations in flow from year to year (Sarr 1995). Summers are typically dry and vegetation productivity becomes moisture limited by late summer, except on riparian sites (Benedict 1983). Diurnal air temperatures vary widely during the season, with daytime temperatures approaching 30° C, and with night frosts common in meadow basins. Summer stream water temperatures are correspondingly variable in the South Fork and its tributaries, regularly ranging from less than 4° C at sunrise to greater than 25 ° C by late afternoon (Knapp and Vredenburg in press).

Research Objectives

The interdisciplinary study presented here compared numerous abiotic and biotic characteristics of stream channel and floodplain habitats inside and outside of cattle exclosures on the Kern Plateau. Based on literature reviews of grazing impacts and stream and riparian recovery (Kauffmann and Krueger 1984, Flieschner 1994), we developed a series of hypotheses of system responses to short-term riparian exclusion. We predicted that the short term exclusion of livestock should result in a series of geomorphological and

abiotic changes, including decreases in stream width:depth ratios (Duff 1979, Platts 1983), decreased bank angles (Behnke and Zarne 1976, Platts 1981), channel and floodplain aggradation due to sediment trapping in emergent and floodplain vegetation at high flows (Elmore and Beschta 1987, Clary and Webster 1990), increased retention and concentration of nutrients in soils, and increased water infiltration rates of floodplain soils (Rauzi and Hanson 1966). We also predicted a series of biotic changes after exclusion from grazing, including greater total vegetation cover on floodplain surfaces, with greater abundance of hydric vegetation (graminoids) versus mesic vegetation (forbs) (Kauffman 1983a, Ratliff 1985, Clary 1995), increased recruitment of willow seedlings (Marcuson 1977, Kauffman 1983b), increased colonization by pioneer plant species, greater macroinvertebrate diversity and abundance, and increased golden trout density and biomass (Platts 1985b).

METHODS

Soils

Soil chemical properties

Soils were sampled inside and outside of the grazing exclosure in Templeton Meadow in 1994. We sampled soils in the five most common near stream habitats, including vegetated stream side (1 m from the water's edge, high *Carex* sp. cover), streamside point bar (gravel deposits, little to no vegetation cover), seasonal channel (topographic lows in which water flows seasonally), wet meadow (dense vegetation cover dominated by grasses and sedges), and sagebrush (dry, dominated by *Artemisia cana*).

At each of 12 stream meanders (six within the cattle exclosure, six outside it), we ran a transect from the stream edge of the inside bend of the meander to 5 m back from the stream, Not every habitat type was found along each transect, but all habitats present were sampled. Soil cores were collected by driving a stainless steel tube (10cm long and 2.5cm in diameter) into the soil, covering the open base with a gardening trowel, and extracting the soil sample. The core samples were placed on ice and transported to the laboratory for analysis. In the lab we measured bulk density, pH, total soil carbon and nitrogen, nitrogen mineralization activity, decomposition activity, and exchangeable nitrogen pool sizes of all samples. Net nitrogen mineralization was assessed using intact core incubations at the study site (Hart et al. 1994). We determined KCl extractable pool sizes within a week of collection, by colorimetric analysis with a LACHAT autoanalyzer.

Soil infiltration

To test the hypothesis that soil infiltration increased as a result of livestock exclosure-we measured the infiltration capacity of cattle trails inside and outside the Templeton exclosure in August 1994. We measured the infiltration capacity of cattle trails outside the exclosure that were currently in use, and of trails inside the exclosure that had not been used since exclosure construction in 1983. We paired trails that were close together, within and outside of the exclosure. Each 'on-trail' test was paired with an off-trail test one meter away. We conducted infiltration tests using half meter diameter ring infiltrometers. We measured falling water head until steady state infiltration rates were reached. Water levels were measured every five minutes.

Geomorphology

Stream physical characteristics

In conjunction with fish population surveys in Mulkey and Ramshaw Meadows, stream physical habitat characteristics were quantified during August, 1994. In Mulkey Meadow, study sites were located 100 m downstream of the exclosure (grazed site), immediately above the downstream end of the exclosure (ungrazed site), and 100 m above the upstream end of the exclosure (grazed site). In lower Ramshaw Meadow, sites were located 100 m below (ungrazed site) and 100 m above (grazed site) the upstream end of the exclosure. In upper Ramshaw Meadow, the ungrazed site was located just inside the lower end of the exclosure and the grazed site was located 100 m downstream of the exclosure. All sites contained 125 m of stream, and at each site we measured stream physical characteristics along each of 25 transects spaced 5 m apart and arranged perpendicular to stream flow (Simonson et al. 1994). At each transect, we measured a series of parameters to quantify general stream channel morphology, including channel width, channel depth, stream width, stream depth, bankfull height, bank overhang, bank angle, bank water depth, and canopy shading. Detailed descriptions of each of these measurements are given in Knapp and Matthews (in press) and are summarized in Table 3. We predicted that channel width:depth and stream width:depth ratios should be smaller and bankfull height and bank water depth should be larger in the absence of grazing as stream channels stabilized, bank angle should decrease and bank overhang should increase in the absence of grazing as banks are tranformed from laid-back to overhung, and canopy shading should increase in the absence of grazing as riparian vegetation recovers.

We used channel width:depth ratio, stream width:depth ratio, bankfull height, bank angle, bank overhang, bank water depth, and canopy shading to produce a single composite geomorphology score ("Geomorphology Index" - GI) for each site. Each parameter was first standardized to a value between 0 and 1 as follows:

$$\text{Standardized value} = \frac{P_i - P_{min}}{P_{max} - P_{min}}$$

where P_i is the parameter value being standardized, P_{min} is the smallest value of P for all sites in a particular meadow and , and P_{max} is the largest value of P for all sites in a particular meadow. A standardized value of 1 always indicated the best condition, and a value of 0 always indicated the worst condition. These judgments on what constituted better habitat were based on recent reviews of the effects of livestock grazing on streams (Platts 1991). The standardized values were then averaged over all parameters for each site to produce a standardized composite score ranging from 0-1 (worst to best stream habitat condition).

Channel Elevation

In July 1994, permanent aquatic habitat transects installed by the Inyo National Forest fisheries crew in summer of 1992 were monitored. The perdendicular transects were spaced at equal intervals along a complete stream meander resulting in four transects per station. Sixteen transects, with eight in excluded and eight in grazed locations, were sampled in Mulkey Meadow. In both Templeton and Monache Meadows, eight transects, four in grazed and four in ungrazed locations were monitored. From a line leveled to permanent benchmarks, we dropped 30 vertical points across the active channel, recording vegetation type and channel depth in the exact locations sampled two years earlier by Inyo National Forest fisheries crews. Ten equidistant point drops were placed within each subsection (e.g. left bank, right bank, and channel) of the transect allowing equal sample numbers for steep or mildly sloping banks. Channel elevations were calculated and compared with the previous sampling for each treatment. We hypothesized that bank and channel aggradation would occur in the absence of livestock grazing. Mean changes in bank and channel elevations over the time period were compared for grazed and protected reaches and analyzed using the paired student t-test.

Lateral erosion

To determine whether lateral bank erosion was different in the presence or absence of livestock grazing, we installed 72 erosion pins inside and outside the Templeton exclosure in August 1993. Locations were placed throughout both grazed and excluded channel reaches to minimize local effects, and pins were placed into lower and upper bank faces flush with the soil surface. Bank erosion rates were determined by measuring pin exposure length in August 1994. Pins which were lost due to excessive erosion were assumed to have eroded one-half the length of the pin. Localized scouring resulting from the presence of the pins was noted in only one case.

Vegetation

Pioneer vegetation

To determine whether pioneer plant species were more abundant inside versus outside grazing exclosures, we sampled these species on newly deposited point bars in Templeton and Ramshaw Meadows in 1985 and 1986 (see Odion et al. 1988 for detailed species descriptions). All plants in 85 0.5 m x 0.5 m plots were identified and counted. In 1985, we used 45 plots inside and 40 plots outside both livestock exclosures. In 1986, the number of plots was increased to 55 inside and 50 outside. Plots were evenly spaced along transects randomly-placed in similar sandbar areas inside and outside the livestock exclosures. All sandbar sites had the following characteristics: 1) Surfaces were newly formed and had less than 5 percent plant cover. 2) Each sandbar had similar substrate texture. 3) Areas sampled were at similar heights (0.1 m-0.2 m) above the late summer water level of the South Fork Kern River. Plant densities in grazed and ungrazed treatments were analyzed using the t-test. Where variances between treatments were unequal, the Mann-Whitney test was employed

Floodplain vegetation cover and composition

In early July, 1993, floodplain vegetation was systematically sampled above, within, and below the livestock exclosures in Templeton and Mulkey Meadows. One pair of sample reaches was located inside and outside the upper exclosure fenceline, and a second pair was located inside and outside the lower fenceline. Each sample reach was approximately 300 m long, a distance selected to contain at least two complete meanders of the stream. Seven transects were placed perpendicular to the floodplain axis at 40 meter intervals in each reach, and vegetation was sampled using a 0.5 x 0.5 m quadrat, placed at 5 m intervals on each transect. We sampled approximately 50 quadrats above, 50 below, and 100 within each of the two exclosures. Percent foliar cover for each species within the quadrat was recorded. Since cattle had not entered the study area, all vegetation cover measurements were of plant communities prior to seasonal grazing. In Templeton Meadow, the edge of the floodplain was typically marked by steeply inclined banks whose toeslopes were approximately 1 m above summer baseflow levels in the South Fork Kern River. In Mulkey Meadow, topographic breaks were usually less clear and transects ended at the point where herbaceous meadow vegetation stopped, typically 1-1.5 m above summer baseflow levels in Mulkey Creek.

Vegetation cover and frequency data from the floodplain vegetation survey were analyzed using paired statistical comparisons (Student t-test, n=2) for the grazed and ungrazed reaches in each meadow. Because of differences in the size and spatial distribution of species, vegetation cover data were converted to importance values using the following formula:

$$\text{SPECIES IMPORTANCE (SI)} = [(\text{percent foliar cover})(\text{percent frequency})]^{1/2}$$

This produced a composite descriptor to equalize large and patchy (*Salix* sp.) and small but ubiquitous species (*Gayophytum* sp.) for a single analysis. These values were considered absolute measures of importance on the floodplain surface. Further, all importance values were divided by the average cover in a given reach to provide a relative index of importance for each species in the existing vegetation community at that location. Quadrat observations were pooled for each treatment in Mulkey and Templeton Meadows allowing two replicates of the exclosure treatment with two grazed controls. Data were tested for significance using the paired student t-test.

All species were categorized as species which have been noted to increase or decrease with the onset of grazing (after Dyksterhuis 1949, Ratliff 1985, and Odion et al. 1990). The exclosure treatment was hypothesized to favor compositional shifts opposite to those observed when grazing is initiated (e.g. "increasers" would decrease in importance and "decreasers" would increase). The accuracy of predicted responses were tested by comparing observed importance values with those predicted by the hypotheses above. The sign test was used to determine the accuracy of predicted community responses (as evidenced by differences in the species relative importance value) in grazed and ungrazed reaches of each meadow individually and for the most common species occurring in both Mulkey and Templeton Meadows.

Willow recruitment

To investigate effects of livestock exclusion on riparian willows (*Salix* sp.), in 1985 we measured willow height, number of meristems, and evidence of grazing damage on naturally occuring willow seedlings along 110 meter reaches inside and outside the upper Ramshaw (see Odion et al. 1988 for further details). To assess the effects of cattle exclusion on seedling recruitment densities over a larger spatial scale, a second survey was conducted in August 1994 during which we counted willow seedlings within two meters of each bank along 125 meter longitudinal transects inside and outside the upper Ramshaw, lower Ramshaw, and Mulkey exclosures. Heights of all willows were measured to the nearest 1 cm. Since the lower Ramshaw and Mulkey exclosures were only three years old at the time of sampling, and seedling densities at the time of exclosure construction were unknown, only individual willows under 40 cm tall were considered new seedlings. This assured that only willows likely to have sprouted since the time of exclosure construction were included in the survey. Seedling densities in grazed and protected reaches were log-transformed and analyzed using ANOVA.

Willow cuttings

In 1983 the original restoration study design included experimental planting of willow cuttings, in conjunction with riparian area fencing. Quarter-mile sections inside, and outside, of Templeton and Ramshaw Meadows were hand-planted in 1982 (by a Forest Service resource crew) and in 1984 (by our study team) with 500 willow stems in each section per planting year. The 'planted' areas were in the downstream half of each fenced area, and in the adjacent unfenced section. All cuttings were taken from mature willow shrubs within the immediate meadow areas, usually from higher gradient locations at the upper or lower portions of the meadow. Willow cuttings were a minimum of 1 cm in diameterand 40 cm long, and were planted with at least 30 cm of the cutting buried with flagging attached to the base at the soil level.

To determine whether the survival of willow cuttings was different in the presence vs. absence of livestock, we surveyed all cuttings that were present inside and outside of fences in both meadows in 1985, and again in 1993. Most cuttings, dead or alive, were still visible in 1985, while only live plants could be reliably counted in 1993. The locations of cuttings on three major floodplain surfaces was also noted to establish whether survival differed among habitat zones: streamside (lowest floodplain terrace: < 0.5m above the baseflow stream surface), upper floodplain (0.5-1.0m above baseflow stream surface, often a gravel bar/bench), and channel bank/upland interface, often at the margin of sagebrush vegetation (>1.0 m

above baseflow stream surface). In 1993 volume of each plant, including naturally-recruited individuals, was estimated by measuring plant diameter at two points, and height at three points.

Aquatic invertebrates

Benthic samples

Aquatic invertebrates are often considered useful biological indicators of water or habitat quality (Resh & Rosenberg 1993). To assist in monitoring changes in aquatic habitat conditions, benthic stream invertebrates were sampled in study reaches of Ramshaw and Templeton meadows before the livestock exclosures were constructed (July 1983), and at 1 year (July 1984) and 10 years (August 1993) subsequent to fencing. We stratified sampling to focus on sand substrates, which generally comprised 75% or more of in-stream habitat in these meadows. Cobble or other large substrates were rare in all reaches and were not included in our original samples.

Benthic invertebrates were sampled using a 30 X 30 cm rectangular Hess-type sampler (ca. 0.1 m^2) with an attached 0.33 mm mesh net to collect organisms from mid-stream substrates. The sampler was forced into the streambed and the substrate disturbed by hand to approximately 10 cm deep to dislodge organisms, which were carried by streamflow into the capture net. Net contents were placed into a bucket, and the organisms plus lightweight organic material elutriated from inorganic sediments by washing six times and pouring each wash into a 0.33 mm aquarium net for subsequent preservation in 70% ethanol. Two such samples were collected at each site and combined into a single sample, with four replicate sites sampled in each experimental section. We sampled five stream sections in Templeton Meadow, an upstream and two downstream controls with two sections within the exclosure fence (one section in each treatment had been originally planted with willow cuttings as an additional treatment), and four sections in upper Ramshaw Meadow, two within the original exclosure fence with upstream and downstream control sections.

In the laboratory, all organisms retained by a 2 mm sieve were identified, to genus in most cases, and measured to the nearest mm under a dissecting microscope. The fine fraction which passed through the sieve was sub-sampled, the first 100 organisms identified and counted, and numbers in the entire sample estimated by extrapolation.

Sediment samples

Sediment samples were simultaneously collected from the same stream reaches to quantify the fine particulate organic matter (FPOM) available to detritus-feeding invertebrates. Samples were collected with a 10 cm-diameter plastic coring tube, and sieved in the field to obtain the fine sediment size fraction (0.125-0.5 mm). The fine fraction was returned to the laboratory, dried at 65° C, and weighed before and after ashing in a muffle furnace to determine FPOM mass. Water samples (2 - 5 l.) were also collected to determine the quantity of organic material transported in the water column, and available to filter-feeding invertebrates. The water was filtered in the field through ashed, pre-weighed GFC filters. Filters were dried for 24 hours at 65° C in the laboratory and weighed before and after combusting all organic material.

Prior to 1994, invertebrate sampling was restricted to sand and gravel substrates because these habitats were considered most representative of general stream conditions. Because other substrates, particularly aquatic vegetation, appeared to contain a different assemblage of invertebrates, we sampled additional aquatic microhabitats found in the three primary study meadows in 1994. These `patch' samples were collected and processed by the same methods as for the sediment samples above. We have no pre-exclusion data for patch species composition, so only post-manipulation comparisons among treatments were made.

Drift samples

To determine the quantity and species composition of invertebrates available to water column-feeding fish inside and outside exclosures, invertebrate drift samples were collected in 1994 in Mulkey Creek. Drift nets were constructed of 0.33 mm mesh attached to 20 x 20 cm wire frames, with retainers for attachment to stationary rods staked into the streambed. On August 5-7, six nets were placed into the stream for 1.5 - 2-hour collection periods at four times during the day (early morning, noon, dusk, and at night). Nets were cleaned frequently to reduce clogging by fine suspended material. Current speeds were measured at the mouths of each net to allow calculation of invertebrate drift densities (Allan and Rousek 1985).

Invertebrates in trout diets

Trout diets were assessed in 1993 by collecting and measuring 25-35 golden trout inside and outside of all four study meadows, including Monache Meadow. In 1994, 20 trout were also collected from below, inside, and above the Mulkey exclosure. Stomachs were removed from all captured fish, and preserved with their contents in 70% ethanol. Invertebrates present in guts collected in 1993 and 1994 were identified to genus and counted under.a dissecting microscope.

Fish Population Characteristics

To determine whether fish population characteristics were different inside and outside of the livestock exclosures, we surveyed fish populations at the same study sites used for stream physical characteristics (below, inside, and above the Mulkey exclosure; inside and outside the upper Ramshaw exclosure). Fish were collected using standard electrofishing depletion techniques (Van Deventer and Platts 1983). To facilitate electrofishing, we divided each 125 m site into five 25 m sections using block nets. We conducted three passes through each section using a Smith-Root Type XII electrofisher producing 400 V with a pulse duration of 90 Hz. The length of time that the electrofisher was running on each pass was similar within a section to ensure a similar electrofishing effort on each pass. Captured fish were measured for fork length to the nearest 1 mm and weighed on an electronic balance to the nearest 0.1 g. Fish were released into the section from which they were captured after the final pass within the section was completed.

The number of fish in each site was estimated from the rate of depletion using maximum-likelihood estimation techniques (Microfish V. 3.0 software; Van Deventer and Platts 1985). The depletion data used in the maximum likelihood estimations were obtained by adding together all fish from the first pass in sections 1-5 (=total number of fish in pass 1), the number of fish from the second pass in sections 1-5 (=total number of fish in pass 2), and the number of fish from the third pass in sections 1-5 (=total number of fish in pass 3) (Van Deventer and Platts 1985). Capture probabilities were similar among size classes (0.59 versus 0.52 for adults [$\geq$-100 mm] and age-0 fish [$\leq$ 55 mm], respectively; paired t-test, N=7, $P>0.2$) and all size classes were pooled for the population estimates. The estimated number of fish per m^2 was calculated for each 125 m site by dividing the estimated number of fish per site by the stream surface area (average stream width x 125 m). The estimated total weight of fish in each section was calculated as follows:

$$\text{est. weight} = \frac{\text{total est. number of fish}^{\wedge} \times \text{(total wt. of captured fish)}}{\text{number of fish captured}}$$

We calculated fish weight per m^2 by dividing the estimated weight per site by the stream surface area. On the basis of a recent review of livestock impacts to stream fish populations (Platts 1991), we predicted that trout density (number/m^2) and biomass (g/m^2) should be greater in ungrazed than grazed areas.

RESULTS

Soils

Soil chemical properties

No clear differences were apparent between grazed and cattle excluded sites for any of the measured soil chemical variables (ANOVA $P > 0.1$). Soils were highly variable in the floodplain and significant differences were foumd among the different habitat types studied (Table 4). Vegetated meadow habitats were biologically active, with high nitrogen and organic content and nitrogen mineralization rates. Sandbar soils were predictably sterile and retained comparatively little biologically. The drier sagebrush habitats were intermediate between sandy areas and mexic meadow sites in most soil attributes.

Soil infiltration

Based on sixty infiltration tests, we found that there was both a significant difference between cattle trails and the general non-trail surfaces, and between in-use trails and relic trails (see Table 5). Non-trail surfaces yielded similar values inside and outside the exclosure. Infiltration rates were lower on trails than off (inside and outside the exclosure), and lower on in-use trails than on relic trails. After observing evidence of past overland flow and surface erosion on cattle trails in Templeton meadow, we had originally theorized that overland flow on compacted livestock trails could be a mechanism of erosion and source of sediment and gully growth. However, infiltration rates are sufficiently high, 0.0025 cm/sec (3.5 in/hr) even on cow trails in this location, that precipitation is unlikely to exceed them and cause runoff. Overland flow might still occur during snowmelt when the ground surface is frozen in which case the trough-like nature of the cattle trails could still concentrate overland flow, resulting in erosion

Geomorphology

Stream geomorphological characteristics

The seven stream geomorphological characteristics differed between inside and outside the grazing exclosures in the predicted directions (Sign test; $P = 0.008$, Table 6). Differences in canopy shading, stream width : depth, bankfull height, and bank water depth were marginally significantly different in all four paired comparisons (Mulkey: inside versus above, inside versus below; Lower Ramshaw: inside versus above; Upper Ramshaw: inside versus below) ($P=0.06$ is the smallest P-value possible with $N=4$), while differences in channel width : depth, bank angle, and bank overhang were not significant. The Geomorphological Index (GI) values were also higher inside than outside exclosures ($P=0.06$).

Channel Aggradation

Active channel transect data showed no differences in rates of channel erosion or aggradation between grazed and protected reaches in Templeton or Mulkey meadows (Table 6, ANOVA $P > 0.25$). Channel erosion was slightly greater than bank erosion in both treatments. In both grazed and ungrazed reaches, mean changes in channel cross-sectional areas indicated continuing erosion, but variation was high.

Lateral erosion

Of the 72 bank pins placed into Templeton Meadow channel walls to measure lateral erosion rates, we were able to measure bank loss associated with 38 pins between 1993 and 1994 (mean loss = 2.8 cm, S.E. 0.7). There was no significant difference in bank erosion inside and outside the Templeton exclosure. Mean bank loss was 2.4 cm (S.E. = 1.1 cm, n = 16 pins) in the presence of livestock, while in the absence of grazing erosion measured 2.8 cm (S.E. = 0.8, n = 22 pins). We did find that during this period more erosion took place on the upper, vertical part of the bank (3.3 cm, S.E. = 0.9, n = 22 pins) than the lower, sloped part of the bank (1.7 cm, S.E. = 0.9, n = 16). In fact, some of the pins in the lower, sloping portion of the bank had been buried, the bank material apparently exfoliating from the vertical face due to wet-dry and freeze-thaw cycles. During a year of high spring runoff, the loose material on the lower bank would be carried away, but during this particular year of low spring runoff, the loose material on the lower bank remained largely in place.

Vegetation

Pioneer vegetation

Disturbance dependent annual forbs dominated the flora of newly-deposited point bars in Mulkey, Ramshaw, and Templeton Meadows, but perennial species were also present. Plants occurring on these sandy streambanks were very small, usually less than 10 cm in height, and cover was sparse, averaging less than 20 percent (Sarr 1995). Characteristic species include *Gnaphalium palustre, Epilobium oregonense, Ranunculus cymbalaria*, and *Alopecuris aequalis* (further details in Odion et al. 1988 and Sarr 1995). Total plant density was significantly greater inside versus outside the Templeton exclosure but not the Upper Ramshaw exclosure in 1985 (Table 7). In 1986, total plant density was higher inside versus outside in both exclosures (t-test $P < .05$). The number of plants increased on the same protected streambank from 1985 to 1986, the third season of protection from grazing, while those outside the exclosures remained steady (Table 7).

Floodplain vegetation cover and composition:

An initial comparison of absolute importance values for both meadows showed significant differences in the abundance of only willows (*Salix* sp.) (Paired t-test p = 0.056, Table 8). Species richness per $0.25m^2$ quadrat was similar between areas inside and outside the Mulkey and Templeton exclosures -(Table 9). Although vegetation cover was similar between areas inside and outside the Mulkey exclosure, plots inside the Templeton exclosure had less vegetation cover than those outside (Table 9).

Relative importance values of individual species were only rarely significantly different inside and outside the Mulkey and Templeton exclosures. *Deschampsia caespitosa* -(Paired t-test, P = .03) was more important inside than outside the exlosures. Willows were also more important inside than outside exclosures. However, because of differences in importance between the two meadows, and low statistical power, relative importance values were not significantly different. In general, herbaceous communities were similar in overall composition in grazed and ungrazed areas of each meadow, but the substantial recruitment of willow seedlings inside relative to outside both exclosures suggests that this riparian component is becoming more important in the absence of grazing.

When relative importance values were compared across the entire plant community, notable differences were observed between areas inside and outside of the Mulkey and Templeton exclosures . The hydric species, *Carex simulata, C. utriculata, and Deschampsia caespitosa* have all been noted to decrease with grazing (Dyksterhuis 1949, Ratliff 1985, Odion et al. 1988), and were all more important inside versus outside of both exclosures (Table 8). Conversely, several opportunistic forb species considered to increase with grazing, including *Arnica mollis, Trifolium longipes,* and *Trifolium*

monanthum, were more important outside the exclosures (Table 8). When comparing species abundances inside and outside the exclosures with those predicted by the literature data for grazed and ungrazed communities (Dyksterhuis 1949, Ratliff 1985, Odion et al. 1988), areas inside of both exclosures both meadows-showed compositional shifts in the direction predicted by recovery from grazing pressure (Sign test, P = .007 and P = .004 for Templeton and Mulkey Meadows, respectively). In addition, when mean values for the 20 most common species in both meadows were compared, all of them showed differences in relative importance values between inside and outside of both exclosures that are consistent with the changes suggested by the literature data (Dyksterhuis 1949, Ratliff 1985, Odion et al. 1988) as indicative of recovery from grazing (Table 10).

Willow recruitment

The 1985 surveys of willows inside and outside the upper Ramshaw exclosure showed depressed height and number of meristems as well as increased evidence of herbivory in the presence of livestock (Figure 3). In addition, we observed that willow seedlings were often associated with herbaceous vegetation (Odion et al. 1989). Based on the 1994 surveys, willow recruitment was substantially greater inside than outside the upper Ramshaw, lower Ramshaw, and Mulkey exclosures (ANOVA, N= 3, P = .008). Densities of willow seedlings in protected areas averaged nearly ten times greater than in adjacent grazed reaches (Figure 4).

Willow cutting survival

In 1985 we found that willow cuttings planted in 1982 and 1984 in upper Ramshaw and Templeton Meadows had highest survival when planted on mid-elevation sites away from the stream edge (Table 11), presumably because scouring and long inundation reduced survival near the water while those planted above the floodplain died from desiccation. Livestock presence also strongly reduced survival of cuttings outside of the exclosure fences (Table 11).

The 1993 resurvey of the planted willows showed that survival, which was fairly low in all sites after approximately 10 years, was greater by a factor of 5 in areas where livestock were exluded (109 vs. 20 surviving plants in Templeton Meadow)(Table 11). Grazing also reduced stature of surviving plants by about one-third (Table 11). Again, cuttings fared best in mid-elevation locations in the floodplain, but those planted near the bank were essentially eliminated by bank erosion outside the exclosure, whereas inside the fences survival in these microsites was substantially greater (Table 11).

As with naturally germinating plants, experimentally planted willows were favored in well vegetated microsites. In 1985, survival on point bars in Templeton Meadow was 50.5% within vegetated microsites and 15.9% on primarily bare substrates. In 1993, 68% of surviving transplants were associated with herbaceous vegetation patches inside the Templeton exclosure while 90% were associated with herbaceous vegetation outside the exclosure.

Aquatic Invertebrates

Benthic densities

While invertebrate densities in the treatment reaches averaged between 100 and 800 individuals per 0.2 m^2 (area of sampler x 2 samples), samples were dominated by organisms less than 2 mm in length, primarily early instars of aquatic insects and microarthropods. Larger organisms (2-3 mm) accounted for only 10 - 20% of total counts, generally with fewer than 80 individuals per sample (Figure 5). Because invertebrate biomass increases exponentially with size, and about one-half of the organisms recovered from trout guts were larger than 2 mm (see Figure 7), we focused on larger invertebrates in our analysis of exclusion effects on aquatic communities.

There were no discernible patterns in invertebrate densities inside versus outside the 10-year exclosures (Figure 5). Prior to exclosure construction in Templeton and upper Ramshaw Meadows, densities of invertebrates >2 mm did not differ significantly between treatment reaches ($F=1.4$, $p=0.09$). In 1984, after one year of livestock exclusion, there were no differences among sites inside versus outside of either exclosure, but in both years densities were slightly, but significantly, lower in Ramshaw than in. Templeton Meadow ($F=4.4$, $p<0.05$).

Ten years following exclosure, there were still no differences in invertebrate densities related to grazing treatment, and invertebrate densities were lower following a relatively high run-off year than in previous years (Figure 5). Analysis of variance of 1994 samples showed that fencing had no effect on benthic densities ($F=0.6$), but that location did have some influence, with greater densities in Ramshaw Meadow as compared with Templeton meadow ($F=5.64$, $p<0.05$).

Although we hypothesized that species diversity-should have increased in response to livestock removal, in 1994 there were no significant differences in mean species richness (mean number of taxa per sample) inside and outside of the upper Ramshaw and Templeton exclosures (Figure 6). Analyzing these data for individual taxa also showed no significant differences inside versus outside either exclosure, although several differed between meadows (Table 12). Larvae of blackflies are semi-sessile suspension feeders, and were more abundant in Ramshaw Meadow than in Templeton meadow (31.7 vs. 12.6 larvae/0.2 m^2 in 1983; 12.1 vs. 1.8 larvae/0.2 m^2 in 1993; $p<0.05$ by t-test for each date, and Table 1). Suspended material in both meadows was composed of 35-50% organic content in both years, and was more concentrated in Templeton than Ramshaw Meadow [8.4 ($\pm$ S.E. 0.7) mg/l vs. 5.2 ($\pm$ 1.0 S.E.) mg/l IN 1983; 32 ($\pm$ 7.0) mg vs. 28 ($\pm$ 6.0) in 1993]; however, blackfly larvae were more abundant in Ramshaw with its lower fine particulate organic matter (FPOM) level than in Templeton with its more unstable substrate.

Baetid and leptophlebiid mayfly nymphs, Chloroperlid stoneflies (*Sweltza*), elmid riffle beetles (*Heterlimnius*), and orthoclad midges are all collecter-gatherers which utilize benthic FPOM as important resources (Merritt and Cummins 1984). All these insects were also more abundant in Ramshaw than in Templeton meadow (Table 1). In 1993 benthic FPOM did not differ significantly between meadows, but tended to be greater in Templeton meadow (52 ($\pm$ 7) mg/core) than in Ramshaw (36 mg ($\pm$ 5 mg SE)/core; N.S. @ $P < 0.05$, t-test), but showed no relationship with treatment areas. Again, this is inversely related to collecter-gatherer abundance patterns, further suggesting that other factors, particularly substrate instability, may be determining insect distributions. Benthic FPOM had not differed among sites in 1983 [42 $\pm$ 6 mg in Templeton meadow vs. 48 $\pm$ 3 mg in Ramshaw; organic content of total fines was 55% vs. 68%, respectively].

Amounts of coarse particulate organic matter (CPOM) collected simultaneously with the benthic samples ranged from 1.3 gr/m^2 to virtually absent in Templeton, Ramshaw and Mulkey Meadows, with a mean of 0.22 $\pm$ 0.15 se gr/m^2. No significant difference was observed among sites, but collections were from midstream, and not from margins where amounts of organic material would likely have differed as a consequence of greater amounts of streamside vegetation in Ramshaw and Mulkey Meadows. `Shredders' and predators (sensu Merritt and Cummins 1984) are fairly rare in these systems, as would be expected in these relatively sterile, shifting habitats. An unexpected contradiction to the general pattern of lower invertebrate abundance in the more degraded Templeton meadow was that two interstitial predators, a crane fly larva (*Hexatoma*) and dance flies (Empididae), were significantly more common in Templeton than in Ramshaw Meadow (Table 12).

Habitat patches

The 1993 benthic samples collected from selected substrate patches showed that invertebrate densities in Mulkey, Ramshaw and Templeton meadows were substantially greater on in-stream and

emergent plants than on more common sand and gravel substrates (Table 13). Densities of large organisms (> 2 mm) mirrored those of total densities. Aquatic vegetation also provided habitat for a greater diversity of invertebrates, with mean taxon richness approximately. 50% greater than on open sand (Table 13). Some taxa largely absent from sand substrates included a dragonfly nymph, *Ophiogomphus*, *Drunella* sp. mayfly nymphs, nemourid stoneflies, alderflies (*Sialis* sp.), *Hydropsyche* caddis larvae, and simuliid blackflies. In contrast, ostracods and elmid riffle beetles were present on both sand substrates and aquatic vegetation.

Although vegetation patches comprised less than 20% of total in-stream substrate area (Table 13), invertebrate densities where more than an order of magnitude greater in vegetation than on the dominant sand substrates, and suggests that these habitats are critical for production of invertebrates, especially Ephemeroptera. Based on our 1993 data there was no significant difference in total stream area occupied by instream vegetation inside versus outside the exclosures (Table 13). There was, however, a greater amount of true aquatic vegetation inside the fence in Templeton Meadow (22% $\pm$ 3% S.E. vs. 12% $\pm$ 2%). mostly owing to more aquatic buttercup (*Ranunculus aquatilis*) in midstream. Despite lack of treatment differences, Mulkey and Ramshaw Meadows sustained greater amounts of channel margin vegetation than did Templeton Meadow (81.5% vs. 71% vs. 58%, respectively), mostly comprised of *Carex utriculata* along the stream edge

Invertebrate drift

The mean drift density at dusk of aquatic insects >2mm was 0.44 organisms/m3 ($\pm$ 0.24 S.E., n=4) in Templeton and 0.34 insects/m3 ($\pm$ 0.27 S.E, n=4) in Ramshaw Meadow. Although terrestrial insects are often very important as food sources, they comprised less than 2% of large drifting organisms (>2mm), and about 3-7% of all drifting organisms in our 1993 samples. Total drift included mostly baetid mayflies (24%), chironomid midge larvae (27%), micro-crustaceans (28%), and aquatic mites (13%).

In Mulkey Meadow, drift patterns were similar, except that drift densities were substantially higher at that time (13-14 July 1994) than they had been in Ramshaw and Templeton meadows (Table 14). Here mayflies were partitioned between two taxa, baetids making up 14%, and 11% of total drift represented by a leptophlebiid mayfly, *Paraleptophlebia* sp. We verified that the bulk of drifting occurs in the evening and at night, during which time densities were approximately 3 to 4 times greater than during the daylight hours (Table 14). Terrestrial insects, however, were more common in the drift at midday and dusk, corresponding to typical periods of diurnal and/or crepuscular activity.

Invertebrates in trout diets

As expected based on the composition of invertebrate drift, trout diets were dominated by mayflies and chironomids (Figure 7). Despite a tendency for fish collected outside of the four exclosures to contain fewer total organisms than fish inside exclosures (13.8 vs. 17.9 items per gut), these differences were not statistically significant ($F=2.66$, $P=0.15$)

These 1993 gut collections had not been differentiated by invertebrate or fish size, but the 1994 sample from Mulkey Meadow were separated by size class. Small organisms <2 mm in length were commonly ingested by these trout (mean proportion 60% of total contents), which were themselves relatively small (mean 13 cm length, with numerous individuals < 8 cm)(Fig. 8). Whereas terrestrial insects comprised about 2% of the total drift in Mulkey Creek, they comprised 30% of trout diets. Common food items included terrestrial flies, wasps, beetles, and other insects that had either fallen into the water or were taken in the air (Figure 8).

Fish Populations

Densities of golden trout (number/m^2) were higher inside than outside exclosures in all four paired comparisons (Wilcoxon signed rank test: N=4, P=0.06), and trout biomass (g/m^2) was higher inside than outside exclosures in three of four paired comparisons (Wilcoxon signed rank test: N=4, P=0.12). In addition, trout biomass was significantly correlated with the geomorphological index (N=7, r=0.83, P<0.03; Figure 9). Detailed descriptions of the differences inside versus outside of exclosures, and additional information concerning the biology and status of golden trout populations in the study area are provided in other reports, including Knapp and Matthews (in press), Knapp and Vredenberg, (in press), Knapp and Dudley 1988).

DISCUSSION

Soils and Geomorphology

Ten years of livestock exclusion did not result in any discernable differences in the soil chemical properties selected in this study. Floodplain terrace soils at the study site are composed of granitic sands and and gravels and skeletal at inception (Albert 1982). Accrual of organic matter appears to be the factor most influential upon soil biological properties and is usually associated with vegetation colonization and growth in Sierra Nevada meadows (Wood 1975, Sarr 1995.). Since total vegetation cover has yet to respond to exclusion, it is unlikely that large scale changes in the soil nutrient properties have occurred at this time.

Some evidence of soil recovery is evident in the changes in infiltration rates seen on cattle trails in the Templeton exclosure. These trails, which were primarily in the dry sagebrush zone, remain poorly vegetated, and our results appear to be due to physical decompression of soils. Although we did not compare trails or surface on the more productive streamside terraces, we predict that they would recover faster because of the additional effects of plant roots and rhizomes.

The physical, chemical, and biological properties of soils have been proposed as key indicators of rangeland health by range scientists (NRC 1994). The ability of healthy soils to capture and retain moisture and nutrients, while resisting erosion, is considered elemental to the productivity and regeneration of rangelands (Smith 1989). Although soil chemical properties, especially organic content, may be excellent predictors of site decline with overuse, they appear respond less quickly to restoration efforts such as exclusion fencing. They represent a notable examples of important environmental factors which are much more responsive to grazing impacts than remediation efforts. In montane meadow ecosystems, losses of soil organic matter, with associated declines in nutrient and moisture retention, may take very long to remedy, because recovery is mediated through vegetation inputs, which are, in turn, dependent on recovery of site hydrology.

Geomorphology

Numerous quantitative and qualitative studies of livestock exclosures have documented adjustments in stream channel morphology, including narrowing of wetted channels, decreased bank angles, and bank stabilization by sod forming vegetation in excluded reaches (Marcuson 1977, Van Velson 1979, Platts 1981, Marlowe et al. 1987). This short-term study showed clear responses in canopy shading and several geomorphological features in Mulkey and Ramshaw Meadows. Templeton Meadow, however, showed no evidence of physical recovery after 10 years of exclusion.

Kondolf (1993) reported that vegetation height and cover responded vigorously to grazing retirement, but channel surface morphology showed no response to 24 years of protection. Emergent and

bankside graminoid vegetation was much taller inside versus outside the exclosures, but increased vegetation growth has not translated into bank or channel aggradation over the short (2 -10 years) time that cattle have been excluded from the study reaches. Several features of wetted channel morphology, however, are responding in Ramshaw and Mulkey Meadows. These two meadows are only lightly incised and are very productive of herbaceous vegetation. Encroachment of emergent *Carex* vegetation into stream channels is clearly evident at both sites and appears be causing the changes in stream morphology.

Established sedge communities are resistant to very high scouring flows in Mulkey and Ramshaw Meadows (R. Knapp personal observation). Observations in the deeply incised Templeton exclosure, however, suggest that the ability of any vegetation to permanently protect geologically unstable banks is limited. Several unofficial surveys of the Templeton exclosure showed impressive vegetation growth on steep channel banks during the drought years of the early 1990s, however, this vegetation was removed by floods in 1993. Clearly, natural recovery of bankside vegetation and channel form are controlled by thresholds of geological stability which varied from site to site in this study. The temporal and structural relationships between graminoid vegetation establishment and stream channel recovery deserve detailed study.

The episodic nature of sediment movement processes along streams further complicates the description of livestock exclusion effects on channel erosion and/or deposition. The seasonal precipition totals for 1993 and 1994 were 160% and 60%, respectably, of mean totals for the southern Sierra Nevada (CADWR, 1994). As measurements were recorded in summer of 1994, flood effects may have masked previous changes between grazed and protected reaches. The 2-10 years of exclusion within the study exclosures may have been insufficient to provide a clear picture of local channel adjustments. Given the extreme variability of flows in the South Fork, a study period of 20 years or more may be required to dampen seasonal variation and document trends in channel and bank erosion/aggradation due to livestock exclusion. Further, livestock effects on basin geology, which can propagate upstream or downstream, probably function on a larger scale than our exclusion treatments. Ideally, comparisons between grazed and protected reaches should be conducted in replicate headwater basins to avoid this complication.

Vegetation

Greater abundances of pioneer species in the absence of livestock pressure indicate a difference in initial successional processes with exclusion. Many of the species which were favored with exclusion are only found in pioneer communities and linkages with later seral communities remain unclear. However, increased cover of pioneer species would be expected to favor local retention of fine textured sediments and plant propagules during low intensity overbank floods. Additionally, several of the species, such as *Potentilla anserina*, *Ranunculus cymbalaria* and *Carex* sp., exhibit stoloniferous or rhizomatous growth and may be important in site stabilization and development. The long-term effects of these initial stages of recovery are dependent upon timing of flood events. For these early seral responses to develop into comparatively stable meadow communities several years with low flows would be required.

The observed differences in plant community structure between grazed and ungrazed areas, while rarely statistically significant for individual species, indicate initial compositional shifts in the direction predicted for recovery by literature data. Hydric sedges and grasses were a greater proportion and opportunistic forbs species smaller proportion of the plant community in ungrazed than grazed areas. This shift toward graminoid dominance in the absence of grazing has been reported by other authors working in mountain meadow systems (Kauffman et al. 1983, Ratliff 1983, Menke et al. 1996). Kauffman (1983) observed that increased litter buildup in ungrazed areas appeared to lower surface temperatures and favor the expansion of hydric species. Mulkey and Templeton Meadows, although structurally different, showed similar responses to protection, especially greater relative importance of beaked sedge (*C. utriculata*) in the absence of grazing. *C. utriculata* is one of the tallest herbaceous species observed at the study site and

shades many of the other species by late in the summer at moist, ungrazed sites. The beaked sedge (*Carex utriculata*) and tufted hairgrass (*Deschampsia caespitosa*) have been described as heavily utilized by grazers but tolerant of moderate grazing (Schultz and Leininger 1990, Menke et al. 1996). On the Kern Plateau, sedge vegetation is the forage most preferred by livestock (D. Sarr pers. observation) and is probably somewhat limited by this pressure under intensive stocking regimes. *D. caespitosa* and *C.utriculata,* two local dominants, are both hardy, however, and appear be quite resistant to moderate use (Sarr 1995).

The trends in species importance observed here lend support to the classification proposed by Dyksterhuis (1949) and modified by other authors working in the southern Sierra Nevada (Ratliff 1985, Odion et al. 1988). Equivocal statistical results for individual species in this study are attributable to differences in the study sites selected, low statistical power, and low magnitude differences in plant abundances inside and outside exclosures. In the Mulkey Meadow study site, the floodplain was well vegetated and stream channel minimally incised. Grazed and ungrazed reaches were similar and present a good comparison of grazed and ungrazed areas. In the Templeton exclosure, however, channel incision and generally low vegetation cover created a relatively disturbance prone floodplain environment compared to the grazed reaches. Vegetation cover was actually lower (25.8% vs. 42.1%) in the exclosure than outside and evidence of both fluvial scouring and deposition greater in both years of the study. Moreover, the vegetation appeared in dense patches within the exclosure, whereas outside it was more thinly but evenly distributed.

The Templeton exclosure was built for resource restoration purposes and placed at a site of maximum channel entrenchment, and presumed vegetation degradation, within the meadow (Tom Dudley, personal observation). The deep entrenchment of the floodplain within the exclosure may seriously retard the rate of floodplain vegetation recovery due to the likelihood of high velocity constrained flood flows. At the Templeton study site, geological controls may presently be of greater importance in controlling total vegetation cover than current grazing levels, although vegetated patches are exhibiting shifts in species composition. Similarly, Myers and Swanson (1992) found after investigating relationships between livestock use and stream erodability in western Nevada, that stream structural type was more important than current livestock impacts in predicting the short term recovery of degraded riparian areas.

This study confirmed that willow seedling growth and establishment are greatly enhanced in protected vs. grazed sites in Ramshaw and Mulkey meadows. Nearby Templeton meadow showed similar but less dramatic trends. Positive willow (*Salix* sp.) responses to grazing exclusion have been documented for other montane stream and meadow systems in the West (Marcuson 1977, Kauffman and Krueger 1983, Schultz and Leininger 1990). Although the increase in densities and relative importance of willows is the clearest response shown by any plant species to grazing retirement, recovery was influenced by site factors. Relatively stable, vegetated sites, with higher soil organic content, and greater nutrient and moisture holding capacity, appear to provide favorable habitat for young willows, especially in the presence of grazing. Short-tem response was, therefore, more dramatic at these sites than at barren, droughty sites in Templeton Meadows. At marginal sites, other authors have described willow recruitment as highly episodic and dependent upon sufficient and well timed precipitation cycles (Sacchi and Price 1992). A clear evaluation of exclusion effects upon willows should include sufficient time for recruitment events to occur on all floodplain surfaces.

Although previous studies have shown that riparian willows are pioneer species which colonize barren fluvial deposits in the absence of herbaceous competition (Harper 1977, Irvine 1979, Ottenbreit 1992, and others), we observed no evidence of willow recruitment inhibition by preexisting herbaceous vegetation at the study sites. Conroy and Svejcar (1991), who monitored willow cuttings planted in the northern Sierra Nevada, observed no increase in willow cutting mortality on areas with dense growth of sedge (*Carex* sp.) compared to barren sites as long as soil moisture was adequate. Willow recruits quickly outgrow the herbaceous vegetation and appear to be quite vigorous even in densely vegetated areas at the study sites (D. Sarr personal observation). *Salix geyeriana* is the most common species of willow found on

large meadows of the Kern Plateau and this species may have unique tolerance for the thick herbaceous competition and anaerobic soils found at those sites.

Grazing removal appears to strongly favor willow recruitment at appropriate sites within the study area and suggests that a woody riparian community may have once been much more widespread than it is currently. The historical extent of the willow populations on the upper Kern Plateau is presently unknown, but current distributions are patchy and largely confined to higher gradient meadows which receive lighter livestock use than the open basins (A. Dennis, USDA Forest Service, unpublished data). It is unknown if the present association of willows with sloping meadows is a true habitat preference or a relict of past land uses. Paleobotanical studies of historical willow distributions in the region, along with further investigations of factors influencing willow recruitment, would be of great help in understanding the extent of natural willow communities in the area prior to livestock grazing.

Willows are considered to be important plants for stabilizing degraded stream channels and restoring riparian communities (Conroy and Svejcar 1991, Skovlin 1984) In our study, tranplanted willow cuttings showed essentially identical responses as natural seedlings to livestock exclusion. Their growth and survival were strongly limited by the presence of livestock and positively associated with herbaceous vegetation patches. The efficacy of riparian restoration using willow cuttings is compromised in the presence of livestock grazing, but cutting survival was low at all locations as was subsequent regeneration. The most promising locations for this restoration method appear to be along the toe-slopes of steep banks in downcut channels. These locations provide optimal moisture and aeration for willow growth (Sarr 1995), and the maturing plants provide protection for bank during high flows.

Aquatic Fauna

Despite a paucity of information on invertebrate responses to livestock grazing (Rinne 1988a), invertebrates have been proposed as important response variables to habitat degradation and recovery (Plafkin et al. 1989) because they are tightly associated with habitat features, and their dispersal and growth are sufficiently rapid to avoid the lag-time that may occur with fish and beneficial plant species. The invertebrate assemblages we recorded are depauperate for both sand-dominated streams (cf. Grimm and Fisher 1989) and semi-arid montane systems (Leland et al. 1989), but the presence of many rare species and other taxa in relatively non-degraded streams in the region (Dudley, unpub. data) suggest that recovery is possible if vegetation and habitat can be improved. Sedimentation, in particular, is a well-known factor reducing diversity and abundance of stream invertebrate assemblages (Lenat et al. 1981), and in combination with loss of stable microhabitat are serious impediments to improving invertebrate productivity and biodiversity in these streams. Although the amount of FPOM available to filterers is quite high in the study streams, larval densities are low compared with many streams (Colbo and Wotton 1981). Numbers were presumably depressed because filter-feeders need stable substrates upon which to attach (Dudley et al. 1986) and these were quite rare in this shifting sand streambed.

The lack of differences in invertebrate assemblages between grazed stream areas and reaches excluded from grazing for over 10 years is likely the consequence of the mobility of stream invertebrates, and scalar relationships between historic impacts and remediation measures. Stream insects are continuously redistributed within the stream channel, primarily by periodic drifting downstream. Thus, organisms found within one treatment area are likely temporary residents in that site and have dispersed via drifting from areas upstream, including areas above fences and subject to impacts of continued grazing there. Even a half-mile exclusion area is unlikely to represent a closed system, and many of the taxa present are known to be regular long-distance drifters (Allan and Russek 1985, Cooper et al. 1993).

Although the exclosures eliminated livestock access to a substantial amount of aquatic habitat, they may have been of an inadequate size to allow recovery of macroinvertebrate communities in a watershed where grazing is ubiquitous. Particularly in the case of naturally unstable and erosive substrates,

continuing degradation outside of fences is likely to result in scouring and downcutting in `protected' reaches downstream, as our geomorphological data earlier suggest. Some elements of recovery documented in Ramshaw and Mulkey Meadows are significant, while Templeton, which was initially more highly incised, has not produced any geomorphological recovery that can be documented. Yet, the lack of recovery response by benthic invertebrates in any site seems to suggest that even in sites where geomorphological changes are occurring, invertebrate communities have yet to respond. In badly degraded systems, recovery may require a larger scale approach, perhaps exclusion of entire meadows, to promote recovery of aquatic invertebrate communities. Longitudinal effects, including invertebrate and substrate mobility, appeared to mask microhabitat preferences for exclosed versus grazed habitats. The use of invertebrates as biological indicators may, therefore, be more suitable when comparing independent sites, rather than adjacent sites, such as these, that have high degrees of physical and biological connectivity.

Our comparisons of golden trout density and biomass show that trout populations, as indicated by unit/ area measurements and biomass, are positively affected by livestock exclusion. In addition, trout biomass was positively associated with geomorphological recovery. These changes, however, are partially offset by decreases in stream width and benthic area as a consequence of stream recovery, such that fish numbers per 125 meter reach remained similar in grazed and in protected areas.

Comparisons of fish populations are generally made based on unit-area measurements (e.g., number/m^2, g/m^2, kg/hectare) (Keller and Burnham 1982; Platts and Nelson 1985b; Beard and Carline 1991; Larscheid and Hubert 1992), but authors have also used unit-volume measurements (e.g., g/m^3; Platts and Nelson 1985a), and unit-stream length measurements (e.g., number/50 m, g/50 m; Rinne 1988). Although unit-area and unit-length measurements are both justified in study designs such as ours, unit-area measurements may provide the more accurate portrayal of exclusion effects. Trout population size is frequently limited by the autotrophic food base (Murphy and Hall 1981; Murphy et al. 1981; Hawkins et al. 1983). Because total autotrophic production should increase with increasing stream width but remain constant on a per area basis, wide stream sections should have a larger total autotrophic food base than narrower stream sections. As a consequence, all other factors being equal, the number of fish per unit stream length should be an increasing function of stream width while the number of fish per unit area should be unaffected by stream width. Therefore, unit-stream length measurements are potentially confounded by effects associated with stream width while unit-area measurements remove these confounds. Based on this reasoning and our results showing that golden trout density and biomass per m^2 were generally greater in ungrazed than grazed sites, we conclude that livestock exclosures provide higher quality habitat, though less of it, than the grazed controls.

Several other studies have also documented positive responses of trout populations to livestock exclusion (Platts 1991). For example, Marcuson (1977) found that the biomass of brown trout, *Salmo trutta*, was more than three times higher in an ungrazed stream section compared to one that was grazed. Platts (1981) reported that fish densities were more than 10 times higher in a stream section subject to light grazing or no grazing relative to a heavily grazed section. Similarly, densities of rainbow trout, *Oncorhynchus mykiss*, and brook trout, *Salvelinus fontinalis*, were higher in an ungrazed versus a grazed stream section (Keller and Burnham 1982). Although these effects are generally attributed to improvements in the quality of physical stream habitat, cumulative effects, such as altered stream discharge regimes and decreased water temperatures are equally important (Van Velson 1979; Platts and Nelson 1989; Li et al. 1994).

The correlation of trout biomass with improved geomorphology suggests that larger golden trout, like other western salmonids, prefer narrow, deep, baseflow channels with adequate protection from ovehanging banks. These changes are occuring with grazing retirement and may be favoring the production of larger fish (see Knapp and Matthews, in press for further details).

Although biomass was significantly associated with geomorphological factors, golden trout populations appear to be remarkably tolerant of extreme environmental conditions. Results of our fish

population surveys show that despite of the impacts of livestock grazing, golden trout exist at extremely high densities in Mulkey and Ramshaw Meadows (1.3 to 2.7 fish/m^2). In comparison with salmonid densities in 277 streams reviewed by Platts and McHenry (1988), these golden trout populations are among the densest reported for trout in the western United States, and are an order of magnitude more dense than the average trout density from all ecoregions in the western United States (0.25 fish/m^2; Platts and McHenry 1988). Because it is unclear whether Platts and McHenry (1988) included age-0 fish in their density estimates, we made the same comparison after removing age-0 fish from our density calculations. Densities of golden trout in Mulkey and Ramshaw Meadows (1.2-2.0 fish/m^2) remained among the densest in the western United States. Biomass of golden trout in our study streams (15.8-21.2 g/m^2) are also among the highest recorded, and are 3-4 times higher than the average trout biomass of streams in the western United States (5.4 g/m^2; Platts and McHenry 1988).

The dramatic rebounds noted in other studies of western salmonid populations have not occured at the Kern Plateau study sites. This may be because the grazed and ungrazed stream reaches provide different, but possibly equivalent, habitat options and golden trout are abundant in both systems. The ungrazed stream reaches are narrower, deeper, more shaded, more productive per unit area, and provide better habitat for adult fish, while the wider grazed reaches provide more area of benthic habitat, as well as wide shallow areas suitable for spawning (R. Knapp unpublished data). The wide, shallow, and open stream channels observed in grazed reaches of the study areas have not proven as disastrous for golden trout as for other western trout populations. Detrimental effects may be mitigated on the Kern Plateau by cold summer nights (so dangerously high water temperatures are of short duration), low levels of aerial predation, and the extreme hardiness and fecundity of the golden trout.

Synthesis and Conclusions

Drawing overall conclusions from the many variables tested is difficult (results for all variables tested are summarized in Table 15). We observed that the elements of the riparian ecosystem reported as responsive to livestock exclusion appear to respond at different rates, and ecosystem recovery is strongly dependent upon site factors. Emergent and pioneer vegetation, willows, trout unit densities, and several elements of baseflow geomorphology tended to respond moderately to short-term livestock exclusion. The stream and immediate streamside environment are the most geologically and biologically active and, not surprisingly, were the most responsive to exclusion. A notable exception was invertebrate communities which showed no clear response. Further away from the baseflow channel, fluvial disturbance events are of greater magnitude, and lower frequency, making measurement difficult to detect over short time periods. We noted no change in recalcitrant elements such as channel elevation, highflow channel morphology, total vegetation cover, and soil fertility. Useful predictions of riparian ecosystem recovery will require a better understanding of how short-term responses, such as demonstrated here, link to larger scale recovery. The elements of the riparian ecosystem display complex causal relationships which change through differing seral states and streamflow levels. If the site is deeply incised, recovery trajectories are further complicated.

Riparian habitats in each of the three study exclosures appear to be recovering. The rate, magnitude, and permanence of change, however, is very different from area to area. In the Mulkey and Ramshaw exclosures emergent vegetation aggressively colonized the stream edge armoring banks, and began shading the stream channel the first year. Conversely, the deeply incised channel in Templeton Meadows remained only partially colonized after 10 years of exclusion. In moderately incised Monache Meadow, another recently constructed exclosure, which was not quantitatively studied because of the lack of a suitable grazed control area, showed more visual evidence of emergent vegetation recovery after two

years than Templeton did after ten. This dramatic recovery rate was probably related to longer growing season and more favorable hydrology at the lower elevation Monache site (D. Hubbs USFS, personal communication). Several unofficial surveys by forest service crews noted substantial vegetation responses in the Templeton exclosure during the drought years of the late 1980s. These advances appeared to be obliterated by the high flows of 1993, indicating that short-term responses do not necessarily translate into ecosystem recovery, especially in unstable channels. Similar flood effects were not noted any of the other exclosures (D. Sarr personal observation).

Given the variability in fluvial geomorphology, flooding disturbance, and vegetation cover observed from site to site in this study, simple predictions of the long term recovery of streamside and floodplain vegetation, aquatic communities, and floodplain morphology are probably inappropriate. A synthesis of livestock exclosure studies such as the recent meta-analysis of livestock grazing effects by Milchunas and Lauenroth (1993) would be helpful in predicting exclosure effects in different fluvial meadow types. A compilation of response timelines for recovery of physical and biotic attributes in characteristic geomorphological sites would be especially useful in helping managers develop accurate predictions for restoration projects using riparian exclosure fencing. Much of the existing research focused on identifying indicators of livestock impacts (NRC1994) would be well complemented with similar indicators of initial succession, resistance, and eventual recovery of riparian ecosystems.

Some comments on the design and application of exclosure studies

In general, livestock exclosures have proven to be useful tools in developing estimates of desired future conditions for long-term range management. Since most of the public rangelands in the West have been grazed continuously throughout their management history, knowledge of pristine (ungrazed) conditions is anecdotal, or lacking entirely. Although most published literature reports a desirable change in selected parameters with livestock removal, interpretation of these findings remains problematic for managers. The dificulties lie in both in the nature of the exclosures themselves and in the studies conducted in them. Flieschner (1994) described two common characteristics of the exclosures used in these studies: (1) their areas are small, usually less that 50 ha; and (2) they have been grazed prior to exclusion. Both of these conditions limit utility of such exclosures for estimating pristine conditions in grazed ecosystems. Determination of livestock impacts in riparian areas are further complicated by the difficulty in separating livestock induced from natural disturbances due to flooding, sedimentation, drought, and native herbivores, phenomena which often function at large spatial scales. Further, individual authors are prone to publish their most dramatic findings and leave the less significant or negative responses unreported. Synthetic treatments, which measure many variables simultaneously at a given exclosure, allow a more comprehensive understanding of the dynamics of ecosystem recovery than singular studies.

The results of this study suggest that ecological responses to grazing retirement are becoming somewhat predictable, but the magnitude and rate of change are quite site specific. Especially in damaged watersheds, the evaluation of riparian recovery will be complicated by the aperiodic controls such as floods. Although ecologists and range scientists will be interested in synthesizing general trends resulting from exclusion studies, local managers will benefit greatly from their own observations of responses (or the lack of them) within the different habitat types of their own administrative regions. Such localized studies, which will be used primarily to provide data for adaptive management in given watershed or ecoregion, will likely come under close scrutiny by opposing sides of the public lands livestock debate. Inferences about livestock impacts in general will continue to be drawn from observations in livestock exclosures.

Although observations from exclusion studies remain an important research avenue, several assumptions inherent in this approach should be considered before attempting to generalize from singular exclusion studies. The following assumptions are especially common: 1. Studies of recovery pathways are suitable ways to garner knowledge about past and present degradational pathways, and have special applicability to grazing management. 2. Recovery of natural floodplain structure and function can occur

within small (several hectares or less) and replicable exclosures. 3. Recovery processes observed in localized reaches can be generalized to reaches in other ecosystems. 4. Long-term exclosures represent examples of historical conditions. Each of these four assumptions have some biological utility, but the limitations of these premises must be considered. For example, a study which shows no evidence of fluvial aggradation after grazing retirement may be interpreted to mean that grazing has no effect upon erosion rates in general. Central to this reasoning is an assumed symmetry of erosional and depositional processes. In many ecological situations such symmetry simply doesn't exist, species extinction being an good example. Ecological alterations resulting from grazing, such as soil compaction, may take much longer to recover than they did to occur. Additionally, exclosed areas large enough to allow recovery of watershed processes are difficult to replicate, for logistical and political reasons, and statistical power is often low as a result. Therefore, the probability of committing a Type II error, or accepting the null hypothesis (e.g. no recovery), when it is false is often substantial. Threfore, lack of responce in a measured parameter between grazed and ungrazed areas following exclusion should not be interpreted as evidence of minimal livestock impacts in grazed areas. For this reason, the authors suggest that managers and scientists supplement information addressing recovery in exclosures, as presented in this study, with studies addressing degradation pathways when considering the effects of current or proposed grazing on ecological systems.

A concept that has attracted the attention of ecologists and,more recently, range scientists is the idea of hysteresis or ecological "memory" of past impacts. General consensus indicates that most public land areas of the West were severely impacted by unregulated livestock grazing which occurred from the latter half of the 1800s into the first quarter of this century (Kauffman and Krueger 1984, Ratliff 1985, Odion et al. 1988, Milchunas and Lauenroth 1993, Flieschner 1994, etc.). While many areas are improving, some show continuing degradation or only minor signs of recovery with lower stocking rates or even livestock removal (Platts 1981, Sanders and Voth 1983, Milchunas and Lauenroth 1993). The lack of short term responses to livestock exclusion can be attributed to two major causes: (1)vegetation type conversions as a result of overgrazing or changed hydrology, (2) disruption of geologic structure and function in an ecosystem due to soil compaction, stream entrenchment, or gully erosion. In riparian meadow ecosystems these phenomena often occur together; the alteration of the structure and hydrology of streamside areas through gully erosion usually results in a change in the streamside vegetation (Figure 10).

When streams draining grazed watersheds become gullied or incised through excess erosion, they are decoupled from their floodplains and three positive feedback loops are put into play, namely, entrapment, impoverishment, and convergence. In entrapment, floods which would normally escape the baseflow channel are "trapped" in the gully and high scouring flows occur. In impoverishment, the floodplain, which would normally be provided with organic debris and fine sediment at relatively short flood intervals, is either deprived of these inputs entirely or inundated with very coarse alluvium at relatively long flood intervals. Either situation produces floodplain terrace substrata that are nutrient and moisture poor and, therefore, inhospitable for most riparian plant species. These substrata in turn are more likely to remain unvegetated and prone to movement during subsequent long interval flood events. In convergence, channel downcutting leads to xerification of floodplains and constriction of palatable riparian species to gully bottoms, resulting in concentration of livestock, and their impacts, into increasingly smaller areas of suitable forage. This increases livestock trampling on steep gully banks and favors further erosion even with low stocking numbers. Even if grazing is excluded from an affected drainage, two of the three pathways set in motion by the initial disturbance will likely remain in effect for some time. Healthy stream/riparian conditions will only be attained after the stream, sediment, and vegetation can re-equilibrate. Time required for such recovery will depend largely upon geographic, geological, and historical variables, and ecologically based timelines for exclusion treatments are badly needed.

Livestock exclosures erected in impacted watersheds may produce dramatic results if the primary cause of degradation is simply livestock herbivory. If, however, the geological processes of entrapment and impoverishment have begun, the results will likely be controlled to a large degree by the severity of these ongoing effects. Even nonincised reaches within grazed watersheds may show slow recovery response to

livestock exclusion if inputs of plant propagules, woody debris, and sediment are sufficiently altered. Studies, therefore, which attempt to describe changes in the form and function of stream and riparian ecsystems after grazing retirement should be discussed with these possible confounding factors in mind. Further, the predictive power of exclosure studies will be helped tremendously if managers anticipate the value of future studies and place exclosures in representative headwater areas with minimum of confounding effects upstream.

A clear lesson from the intensive study of three, and repeated field observations of four more, newly exclosed areas of the Kern Plateau, is that generalizations that can be made from each exclosure would be limited. Each of the three areas appears to be recovering, especially with regards to streamside vegetation and geomorphology. Recovery responses vary greatly from area to area. Elevation, stream discharge, distance to propagule sources,and channel form appear to be key factors in predicting ecosystem recovery in meadows of the Kern Plateau. Further work on prioritizing the effects of each of these factors would be very helpful in gaining insight into the dynamics of recovery in these and other riparian meadow ecosystems.

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Figure Captions

Figure 1. Gully erosion in Freckles Meadows, Inyo National Forest (circa 1930s). Note rider on horseback in gully.

Figure 2. Map of meadows, exclosures, and study sites in the Golden Trout Wilderness. Dashed lines indicate meadow borders, and solid bold lines indicate grazing exclosures.

Figure 3. Mean size as height and number of meristems (± 1s.e.) and evidence of grazing damage of naturally ooccurring willow (*Salix*) seedlings inside (n=17) and outside (n=13) of the Ramshaw Meadows exclosures, 1985. All plants were within a 110 m length of stream channel. Plant size differences were significant ($P < 0.05$).

Figure 4. Willow (*Salix*)seedlings (plants < 40 cm tall) counted along a 2 x 125 m belt transect in grazed and excluded areas, Ramshaw and Mulkey Meadows, 1994.

Figure 5. Mean benthic densities of large invertebrates (> 2 mm) from inside and outside exclosures in Ramshaw and Templeton Meadows. Double hatching represents 1983 collections, solid bars are 1993, and single hatched bars are collections in 1994. Sites are along x-axis,with Templeton followed by Ramshaw sites; 1,4, and 5 are from grazed sites and 2 and 3 are within fences, with sites 2 and 3, and 4 and 5, combined for Ramshaw Meadow. Error bars are ommitted for clarity, with statistical analyses described in text.

Figure 6. Mean taxon richness for benthic samples (all size classes) from Ramshaw and Templeton Meadows. Sites and dates as in Figure 5.

Figure 7. Mean number of food items (± 1 s.e.) in trout guts collected inside and outside of livestock exclosure fences in 1993 from Mulkey, Ramshaw, and Templeton Meadows. aqin=aquatic insects, aqsm=aquatic inverts < 2 mm, totaq=total aquatic invertebrates, terrsm/terrlg=terrestrial invertebrates < 2 mm and > 2 mm, elm=elmid larvae and pupae, eph=all ephemeroptera, sim=simuliium.

Figure 8. Mean number of food items (± 1 s.e.) in trout guts from Mulkey Meadows in stream areas excluded from (site 1) or subject to (site 2) livestock grazing in 1994. aqins=aquatic insects, peraq=percent of items that are aquatic (as opposed to terrestrial) insects, persm=percent that are < 2mm, biomass=approximation of invertebrate biomass, ml=mean fish length in each sampling area, full=ml/biomass (a measure of fullness), baet=baetis, para=paraleptophlebia.

Figure 9. Relationship between the Geomorphological Index and golden trout biomass (g/m^2). The correlation is statistically significant.

Figure 10. Cross-sectional diagram showing floodplain morphology, water table, and plant community distribution at a site with an incised and a nonincised floodplain (from Sarr 1995).

Figure 1.

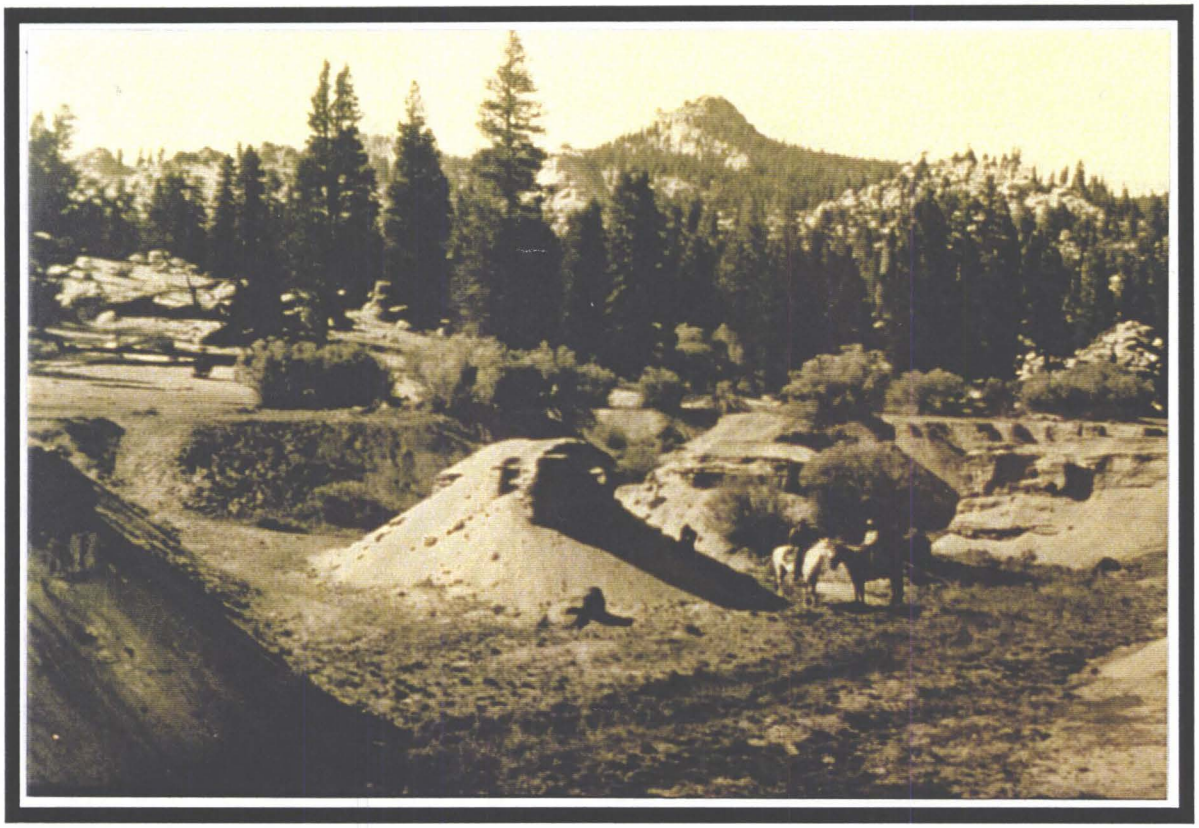

Figure 2.

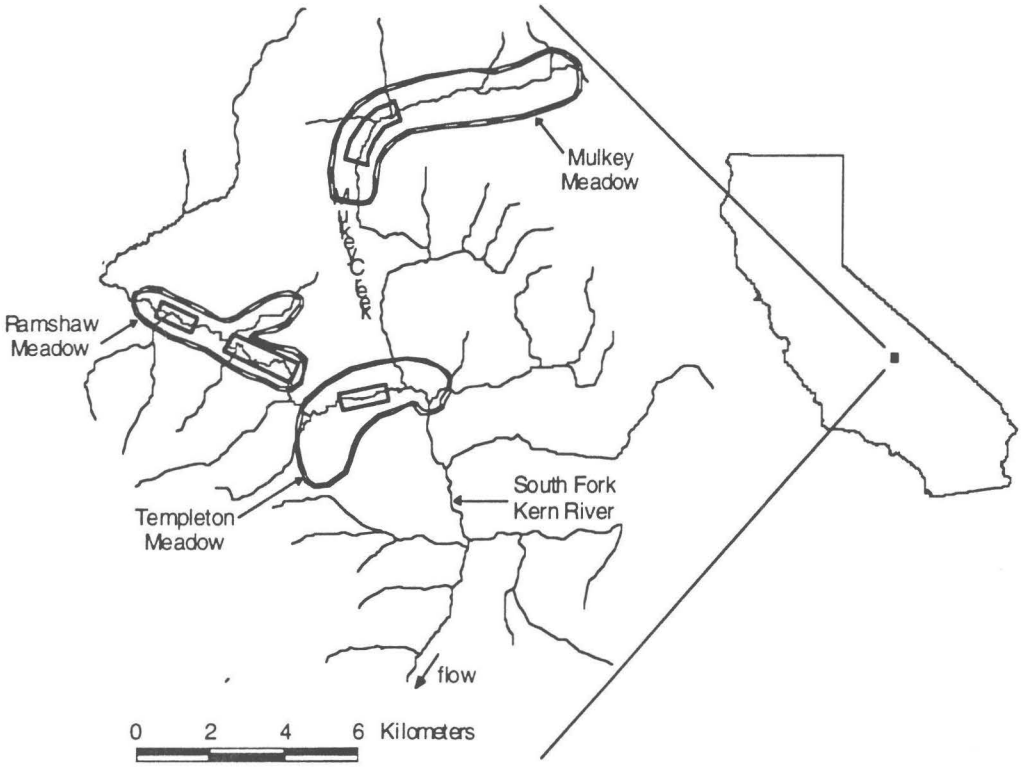
Mulkey
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Figure 3.

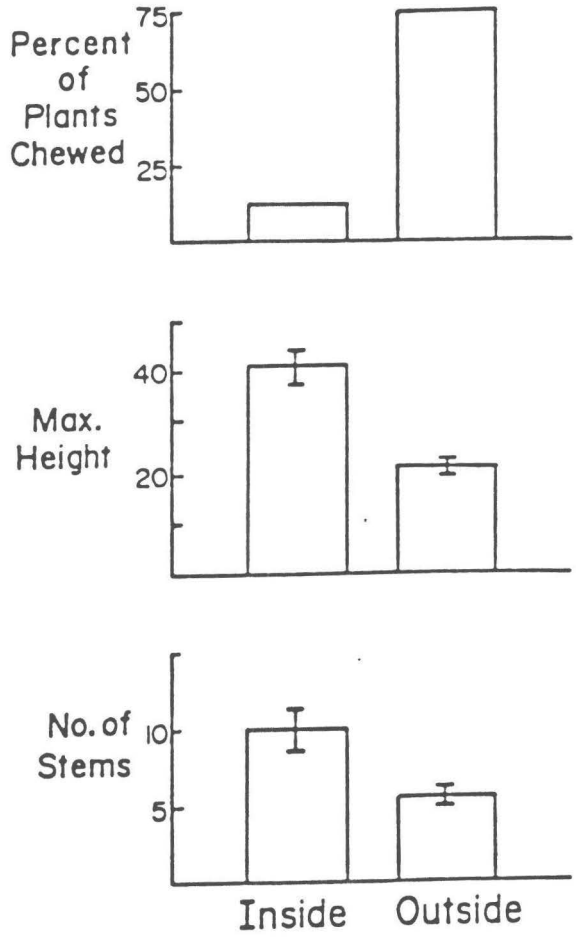

Percent of Plants Chewed
75
50
25
Max. Height
40
20
No. of Stems
10
5
Inside
Outside

Figure 4.

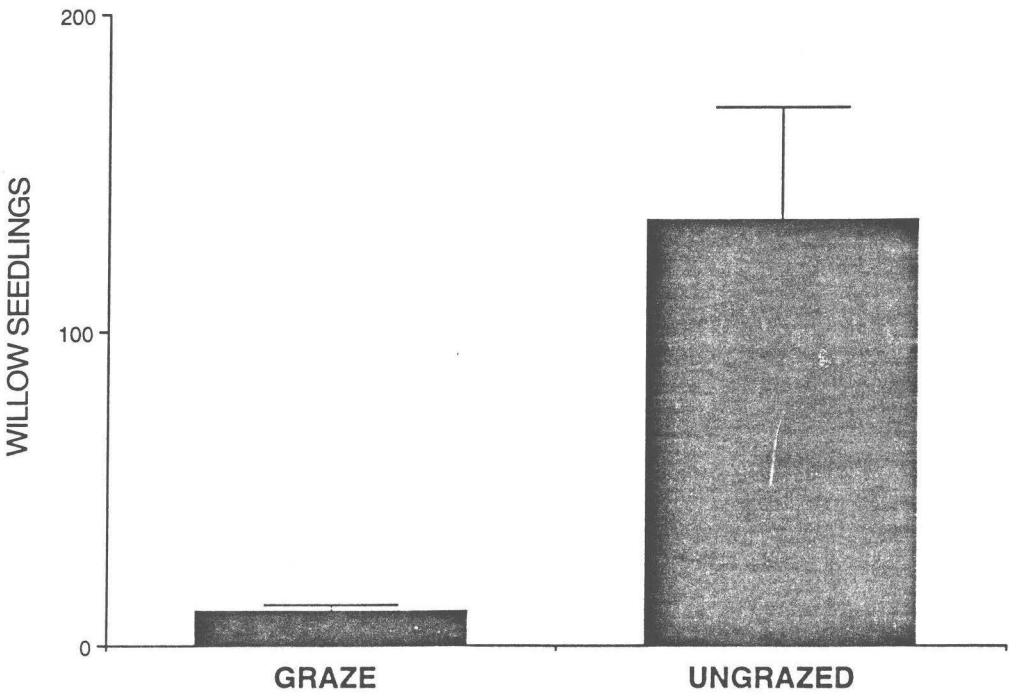
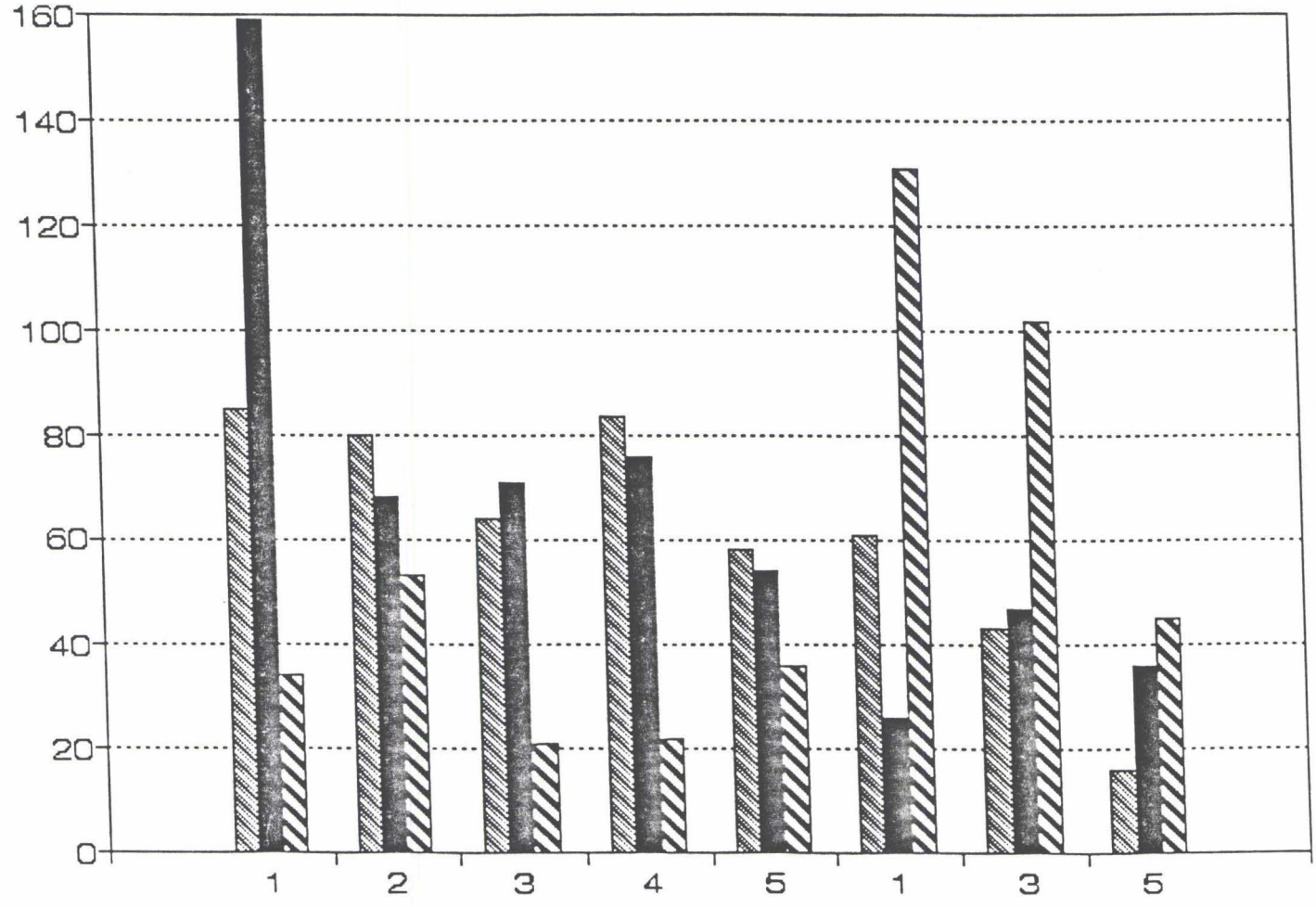


Figure 5.



gure 6.

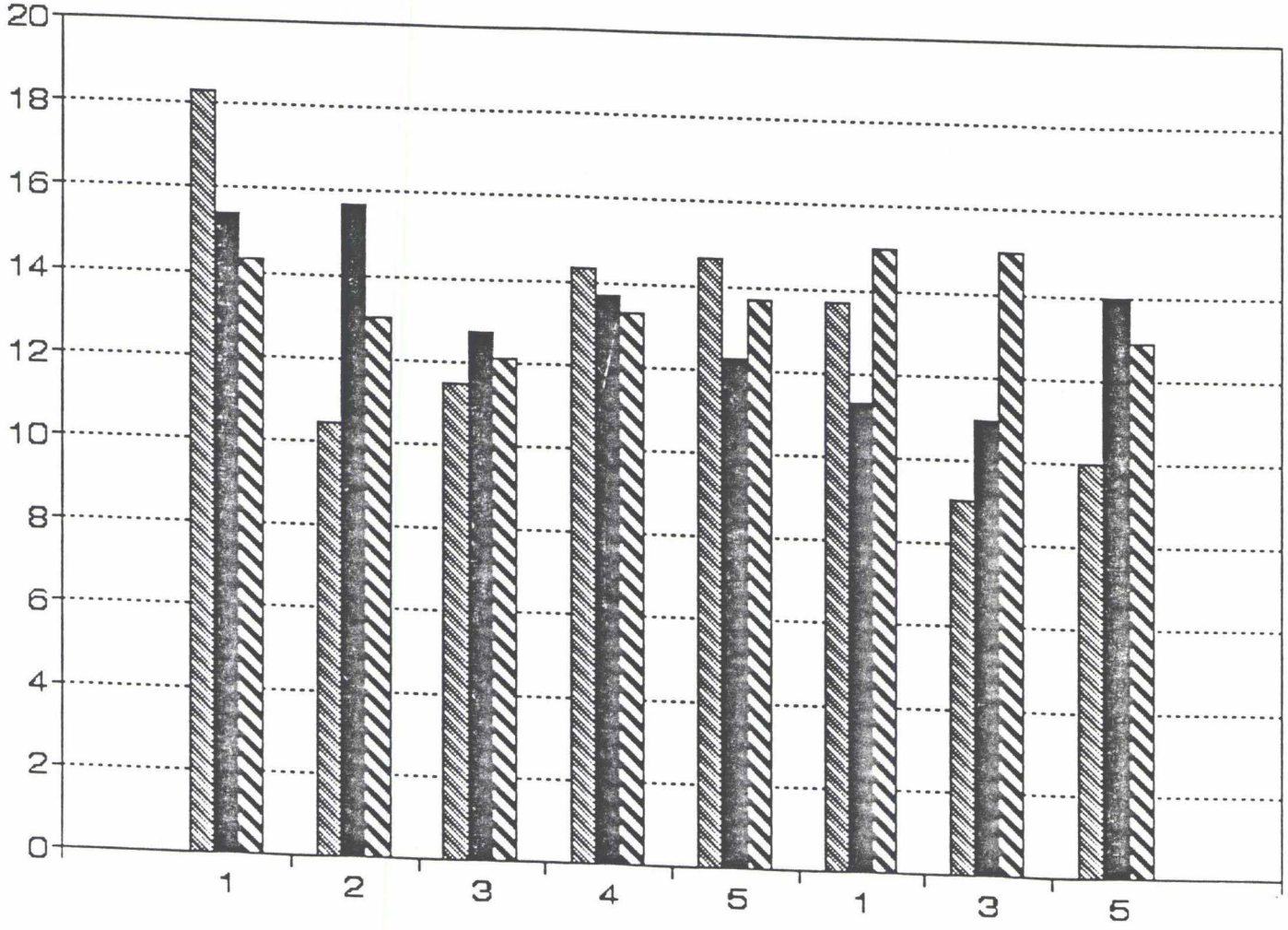


Figure 7.

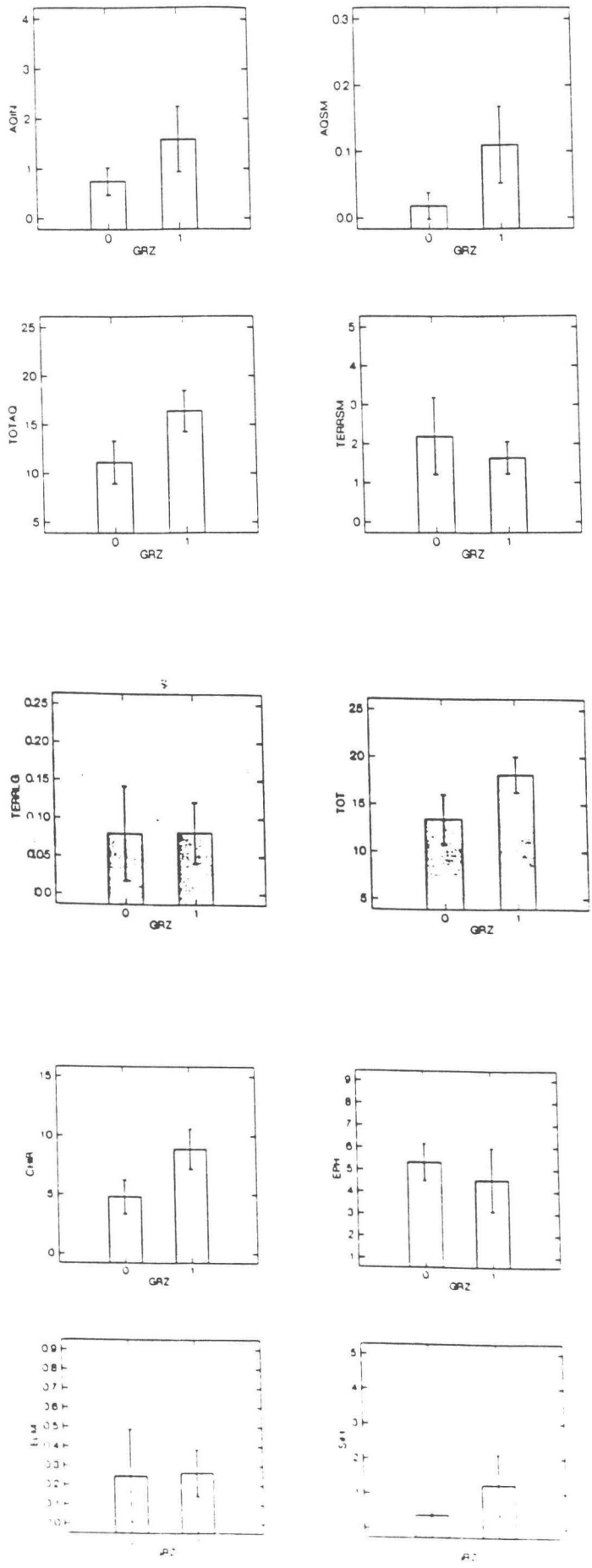
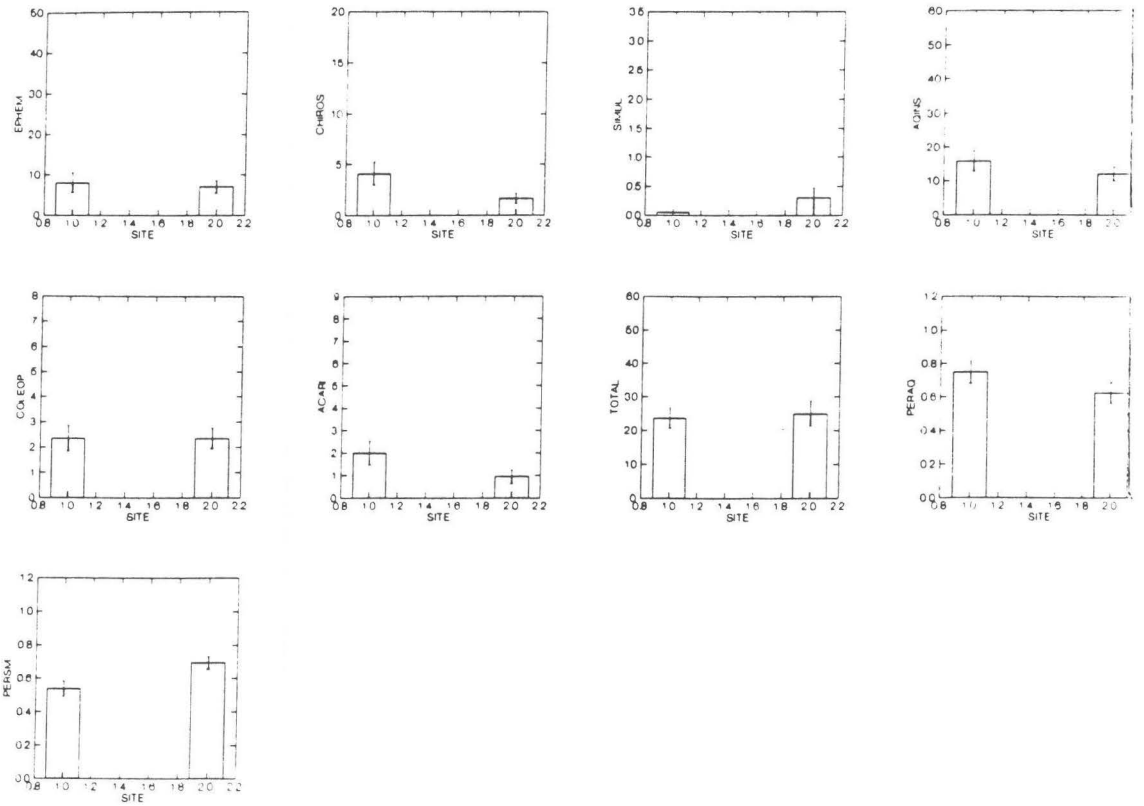


Figure 8.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CHIROS
SIMUL
AQUIS
COLEOP
ACAR
TOTAL
PERAQ
PERSM
SITE

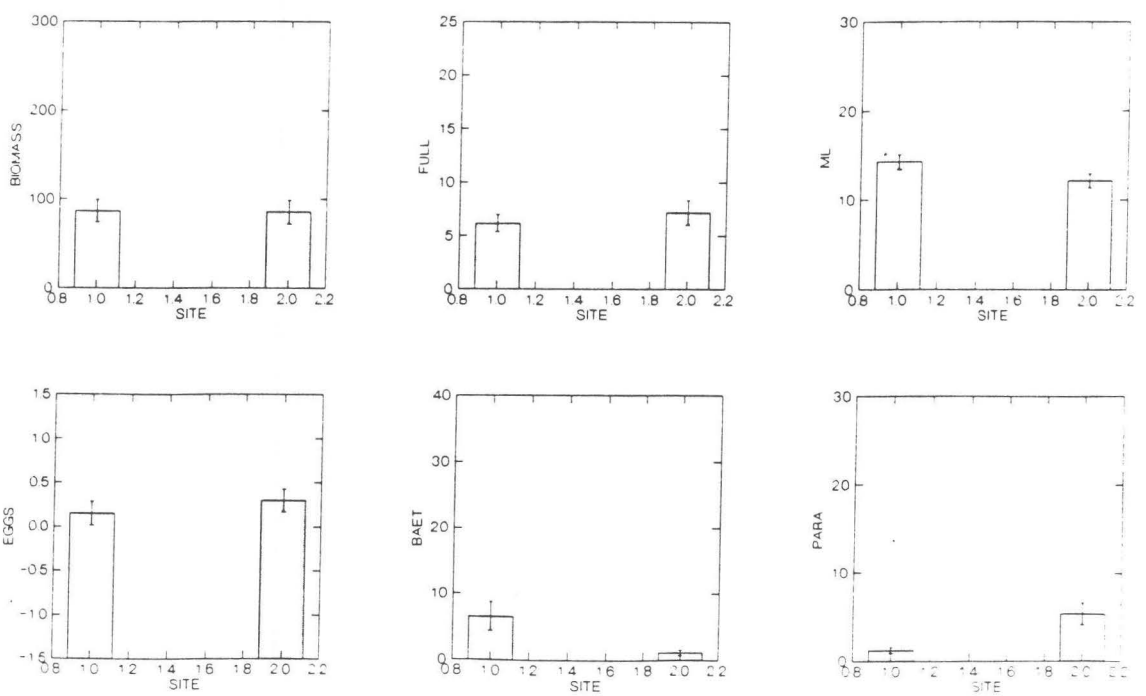
BIOMASS
FULL
ML
EGGS
BAET
PARA
SITE

Figure 9.

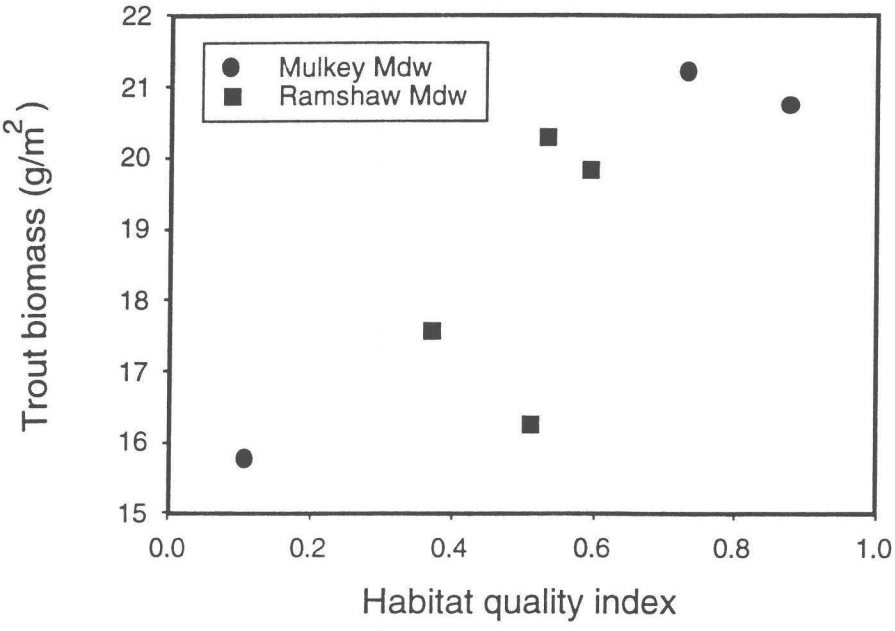
Mulkey Mdw
Ramshaw Mdw
Trout biomass (g/m2)
Habitat quality index
22
21
20
19
18
17
16
15
0.0
0.2
0.4
0.6
0.8
1.0

Figure 10.

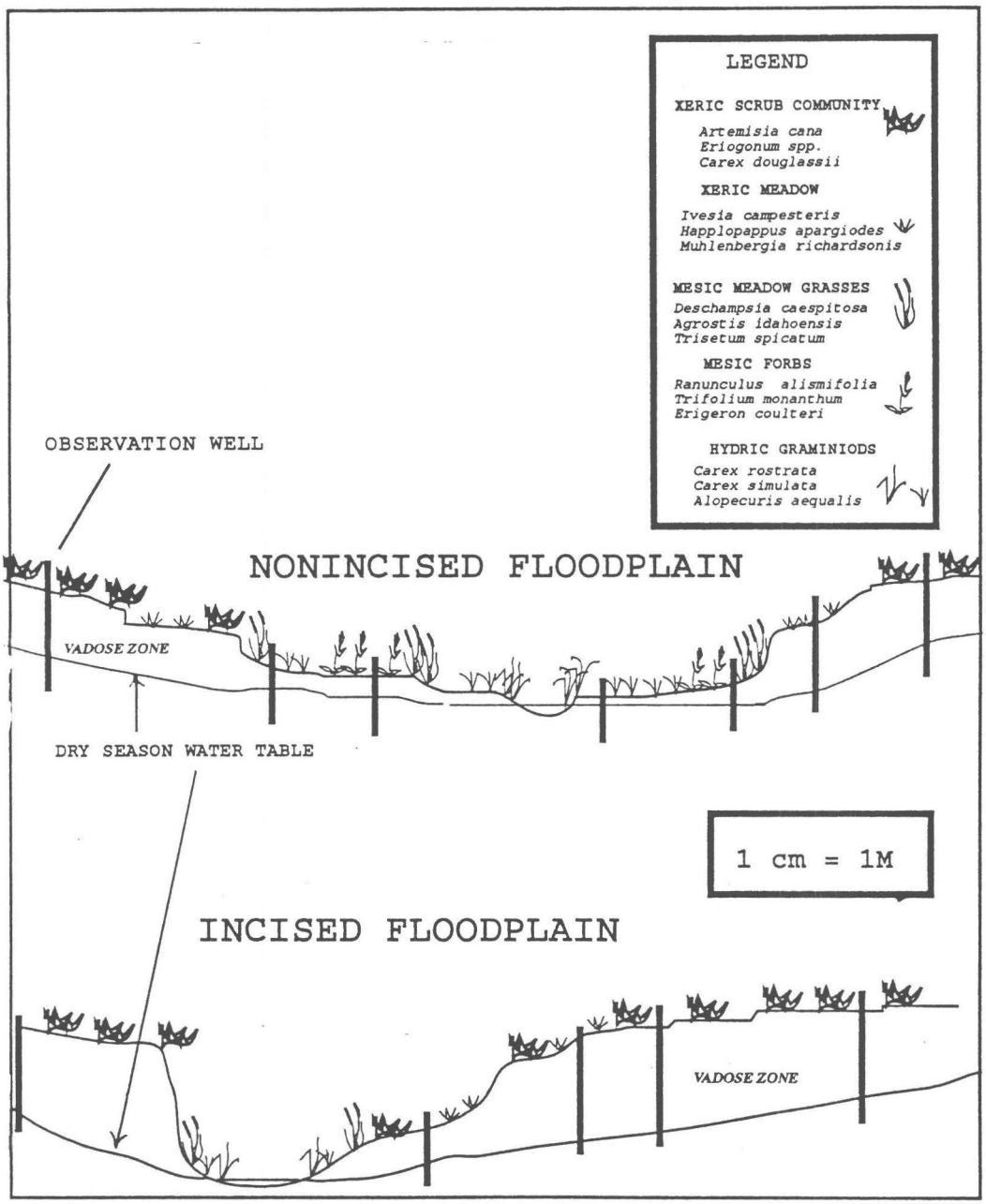
LEGEND
XERIC SCRUB COMMUNITY
Artemisia cana
Eriogonum spp.
Carex douglassii
XERIC MEADOW
Ivesia campesteris
Happlopappus apargiodes
Muhlenbergia richardsonis
MESIC MEADOW GRASSES
Deschampsia caespitosa
Agrostis idahoensis
Trisetum spicatum
MESIC FORBS
Ranunculus alismifolia
Trifolium monanthum
Erigeron coulteri
HYDRIC GRAMINIODS
Carex rostrata
Carex simulata
Alopecuris aequalis
OBSERVATION WELL
NONINCISED FLOODPLAIN
VADOSE ZONE
DRY SEASON WATER TABLE
1 cm = 1M
INCISED FLOODPLAIN
VADOSE ZONE

Table 1. Exclosures studied in Ramshaw, Mulkey, and Templeton Meadows by age and type.

MEADOW	EXCLOSURE	TYPE	YEAR CONSTRUCTED
TEMPLETON	UPPER-T1	BARBED WIRE	1983
	LOWER-T2	ELECTRIC	1991
MULKEY	M1	ELECTRIC	1991
RAMSHAW	OLD-R1	BARBED WIRE	1983
	NEW-R2	ELECTRIC	1991

Table 2. Physical characteristics of the study meadows. Stream order is the Horton stream order estimation; sinuosity is calculated as the ratio of stream channel length : linear transect distance; other parameters are self-explanatory.

Parameters	**Mulkey**	**Ramshaw**	**Templeton**
Elevation (m)	2900 - 2810	2700 - 2660	2680 - 2630
Meadow Area (ha^2)	234	390	727
Drainage Basin Area (km^2)	22.0	38.8	71.8
Stream Order	3^{rd}	4^{th}	4^{th}
Gradient (% slope)	1.0 - 2.5	0.8 - 1.2	1.0 - 1.75
Channel Incision (m)	0.3 - 4.0	0 - 2.4	1.0 - 3.5
Baseflow Discharge (m^3/sec)	0.2 - 0.35	1.2 - 1.5	1.3 - 2.4
Sinuosity	1.4 - 1.6	1.3 - 1.8	1.3 - 2.4

Table 3. Descriptions of measured stream geomorphological parameters.

Parameter	Description
Channel width	Cross-section containing the stream that is distinct from the surrounding area due to breaks in the general slope of the land, lack of upland vegetation, and changes in the composition of substrate materials. One measurement per transect.
Channel depth	Distance from the top of the channel to the water surface. One measurement per transect.
Channel width : channel depth	Average channel width divided by average channel depth.
Stream width	Width of the present water surface not including islands. One measurement per transect.
Stream depth	Average water depth as measured at five equally-spaced points across each transect.
Stream width: stream depth	Average stream width divided by average water depth.
Bankfull height	Height above the current water level at which the banks lose their ability to contain the stream. Measured at both banks along each transect.
Bank angle	Bank slope measured using a clinometer. Overhanging banks have angles of 0-<90°, and laid-back banks have angles of 90-180°. Measured at both banks along each transect.
Bank overhang	Distance from deepest bank undercut to the furthest point of bank protrusion. Measured at both banks along each transect.
Bank water depth	Water depth 15 cm from the stream bank. Measured at both banks along each transect.
Canopy shading	The amount of shading cast onto the stream surface by riparian vegetation. At each transect, measured using a densiometer at both banks and in midstream facing up and downstream.

Table 4. Soil chemical and physical parameters

Habitat	pH	bulk density (Mg m^{-3})	Carbon (g kg^{-1})*	Nitrogen (g kg^{-1})*	C/N	sample size
Artemisia	4.50a 0.06¥	1.35a 0.06	25a 3.3	1.91a 0.22	13.1	12
Point Bar	5.62b 0.15	1.87b 0.04	2.4b 0.41	0.23b 0.27	10.5	8
Meadow	4.46a 0.03	1.02c 0.05	42a 12.4	3.08a 0.77	13.5	5
Seasonal Channel	5.34b 0.13	1.37a 0.08	6.6b 0.88	0.52b 0.18	12.6	8
Streamside	4.46a 0.07	1.36a 0.06	9.4b 0.30	0.67b 0.18	14.0	12

means followed by the same letter are not significantly different ($p<0.05$)
¥ standard error of the mean

Table 5. Results of soil infiltration tests inside versus outside the livestock exclosure in Templeton Meadow.

Group	n	infiltration mean value (cm/sec)	±	standard error (cm/sec)
Trail	30	0.0032	±	0.0003
Non-trail	30	0.0069	±	0.0005
In use trails	15	0.0025	±	0.0003
Relic trails	15	0.0038	±	0.0004
Non-trail, Outside Exclosure	15	0.0067	±	0.0006
Non-trail, Inside Exclosure	15	0.0071	±	0.0009

Notes:
"n" is number of individual infiltration measurements (sample size)
"standard error" is the standard deviation divided by the square root of "n"
in order to compare infiltration rates to typical rainfall values:

0.0032 cm/sec =	4.5 in/hr
0.0069 cm/sec =	9.8 in/hr

Table 6. Mean (and s.e.) values of erosion and stream geomorphological charactaricstics measured in grazed and protetcted areas of Templeton, Ramshaw, and Mulkey Meadows, 1994. T=Templeton, R=Ramshaw, and M=Mulkey Meadows

Geomorphology Results						
PARAMETER	Parameter value		Percent difference	Meadow(s)	Statistical Test	Result
	Grazed	Ungrazed				
Channel and bank erosion (m)	.022 (.016)	.045 (.064)	105	T,M	paired t-test	N.S. (P<.25)
Canopy shading	13 (5)	29 (4)	123	R,M	WILCOXEN	0.06
Bank Angle	131 (100)	119 (4)	-10	R,M	WILCOXEN	> 0.1
Channel width:depth	14 (2)	10 (2)	-40	R,M	WILCOXEN	> 0.1
Stream width:depth	20 (5)	14 (3)	-43	R,M	WILCOXEN	0.06
Bankfull height (cm)	15 (1)	22 (3)	47	R,M	WILCOXEN	0.06
Bank overhang (cm)	4 (2)	6 (1)	50	R,M	WILCOXEN	> 0.1
Bank water depth (cm)	11 (2)	17 (2)	82	R,M	WILCOXEN	0.06
Geomorphological Index (GI)	0.43 (0.16)	0.72 (0.13)	67	R,M	WILCOXEN	0.06

Table 7. Mean number of plants (± 1 s.e.) in 1/2 x 1/2 m plots on sandbars inside vs. outside of livestock exclosures in major meadows of the South Fork Kern River after 2 and 3 years of exclusion; * = significant difference by t-test at $p < 0.01$. From Odion et al. (1988).

		INSIDE			OUTSIDE	
		n	Abundance		n	Abundance
RAMSHAW	1985	25	26.6 ± 2.7		20	21.0 ± 2.0
	1986	20	44.9 ± 4.2	*	20	15.4 ± 2.9
TEMPLETON	1985	25	52.9 ± 6.9	*	30	22.8 ± 3.2
	1986	30	75.3 ± 7.2	*	30	28.1 ± 3.9

Table 8. Absolute and relative importance values for common plant species in grazed and ungrazed areas with results of paired Student-T tests between grazed and ungrazed sites, Templeton and Mulkey Meadows.

MEADOW	MULKEY				TEMPLETON					
	INSIDE		OUTSIDE		INSIDE		OUTSIDE		T-TEST	T-TEST
	absolute	relative	absolute	relative	absolute	relative	absolute	relative	abs.	rel.
GRASSES										
Agrostis idahoensis	6.29	9.46	5.23	7.83	0.50	1.94	1.99	4.71	N.S.	N.S.
Hordeum brachyantherum	0.00	0.00	0.93	1.39	0.97	3.76	3.72	8.83	N.S.	N.S.
Poa pratensis	0.52	0.78	2.72	4.07	1.62	6.28	3.17	7.51	N.S.	N.S.
Torreyochloa pallida	1.87	2.81	1.36	2.03	3.75	14.53	4.33	10.28	N.S.	N.S.
Sitanion hystrix	1.01	1.52	1.19	1.79	0.00	0.00	0.00	0.00	N.S.	N.S.
Muhlenbergia filiformis	2.85	4.28	0.88	1.32	0.38	1.46	0.00	0.00	N.S.	N.S.
Muhlenbergia richardsonis	17.56	26.42	20.90	31.30	3.79	14.68	6.80	16.15	N.S.	N.S.
Elymus trachycaulis	0.20	0.30	0.16	0.23	0.22	0.84	0.00	0.00	N.S.	N.S.
Alopecuris aequalis	0.20	0.30	0.20	0.30	0.43	1.68	0.00	0.00	N.S.	N.S.
Deschampsia caespitosa	8.25	12.42	6.39	9.58	3.21	12.45	4.11	9.75	N.S.	p =.03
Trisetum spicatum	4.86	7.31	3.68	5.51	0.00	0.00	0.00	0.00	N.S.	N.S.
Phleum alpinum	0.46	0.70	0.51	0.76	0.00	0.00	0.16	0.37	N.S.	N.S.
Calamagrostis stricta	0.00	0.00	0.00	0.00	0.40	1.54	0.00	0.00	N.S.	N.S.
Poa stebbensii	0.28	0.43	1.14	1.71	4.97	19.28	2.99	7.10	N.S.	N.S.
Poa nevadensis	1.47	2.22	1.85	2.76	0.00	0.00	0.00	0.00	N.S.	N.S.
GRAMINOIDS										
Carex douglasii	0.35	0.52	0.81	1.21	1.52	5.91	0.47	1.12	N.S.	N.S.
Carex utriculata	4.51	6.78	3.32	4.97	3.32	12.86	3.55	8.43	N.S.	N.S.
Carex simulata	13.93	20.95	10.22	15.31	4.54	17.60	5.74	13.63	N.S.	N.S.
Carex athrostachya	2.51	3.77	2.18	3.27	2.09	8.11	2.79	6.63	N.S.	N.S.
Carex nebraskensis	0.00	0.00	0.09	0.13	1.47	5.72	0.00	0.00	N.S.	N.S.
Juncus balticus	2.08	3.13	4.38	6.56	0.09	0.34	1.11	2.62	N.S.	N.S.
Juncus orthophyllus	1.43	2.15	2.13	3.19	0.00	0.00	0.50	1.18	N.S.	N.S.
Eleocharis macrostachya	0.35	0.52	0.00	0.00	0.00	0.00	0.00	0.00	N.S.	N.S.
Eleocharis pauciflora	1.02	1.53	1.71	2.56	1.66	6.43	4.20	9.96	N.S.	N.S.
Eleocharis acicularis	9.33	14.03	14.79	22.16	0.00	0.00	0.00	0.00	N.S.	N.S.
SHRUBS										
Artemisia cana	22.38	33.68	22.11	33.13	5.78	22.39	13.15	31.21	N.S.	N.S.
Salix spp.	1.94	2.91	0.13	0.19	2.16	8.36	0.00	0.00	p = .04	N.S.
FORBS										
Arnica mollis	2.63	3.96	14.75	22.10	1.06	4.11	1.70	4.04	N.S.	N.S.
Arnica sororia	2.95	4.44	4.89	7.32	2.12	8.22	2.93	6.96	N.S.	N.S.
Castilleja nana	0.00	0.00	0.30	0.45	0.00	0.00	0.08	0.19	N.S.	N.S.
Chryptantha circumscissa	0.00	0.00	0.07	0.10	1.17	4.53	1.82	4.33	N.S.	N.S.
Gayophytum decipiens	1.70	2.56	2.06	3.09	4.48	17.36	8.41	19.95	N.S.	N.S.
Gayophytum decurrens	0.00	0.00	0.66	0.99	0.00	0.00	0.44	1.06	N.S.	N.S.
Eriogonum spergulinum	0.00	0.00	0.46	0.69	0.00	0.00	0.00	0.00	N.S.	N.S.
Aster occidentalis	2.13	3.21	3.76	5.64	1.08	4.20	2.05	4.86	N.S.	N.S.
Erigeron peregrinus	6.41	9.65	8.35	12.51	0.00	0.00	0.61	1.44	N.S.	N.S.
Lupinus breweri	1.01	1.52	0.00	0.00	2.12	8.22	0.67	1.58	N.S.	N.S.
Antennaria rosea	2.33	3.51	1.59	2.38	0.00	0.00	0.00	0.00	N.S.	N.S.
Ivesia campestris	4.54	6.83	5.01	7.51	0.53	2.05	1.06	2.52	N.S.	N.S.
Haplopappus spergiodes	1.34	2.01	4.43	6.64	0.28	1.09	2.10	4.97	N.S.	N.S.
Stellaria longipes	0.35	0.52	0.59	0.88	0.00	0.00	1.28	3.03	N.S.	N.S.
Potentilla anserina	0.54	0.81	0.97	1.45	3.67	14.24	6.47	15.34	N.S.	N.S.
Potentilla gracilis	0.46	0.70	0.95	1.43	0.00	0.00	0.00	0.00	N.S.	N.S.
Mimulus primuloides	2.56	3.86	1.96	2.94	0.18	0.70	0.63	1.49	N.S.	N.S.
Penstemon heterodoxus	0.25	0.38	1.35	2.02	1.22	4.75	0.74	1.75	N.S.	N.S.
Epilobium oregonense	0.22	0.33	0.38	0.57	0.18	0.70	0.31	0.72	N.S.	N.S.
Trifolium monanthum	9.17	13.79	9.87	14.78	0.59	2.28	2.08	4.94	N.S.	N.S.
Trifolium longipes	0.66	0.99	3.27	4.90	6.03	23.36	10.27	24.38	N.S.	N.S.
Gnaphalium palustre	0.00	0.00	0.29	0.44	1.14	4.44	0.43	1.03	N.S.	N.S.
Ranunculus alismifolia	5.13	7.72	5.15	7.71	0.00	0.00	0.00	0.00	N.S.	N.S.
Dodecatheon alpinum	4.56	6.86	2.90	4.35	0.43	1.68	0.35	0.84	N.S.	N.S.
Gentiana holopetala	3.35	5.04	2.16	3.23	0.00	0.00	0.00	0.00	N.S.	N.S.
Gentiana newberryi	1.53	2.30	0.40	0.60	0.00	0.00	0.00	0.00	N.S.	N.S.
Polygonum polygaloides	2.11	3.18	0.52	0.77	0.22	0.84	0.00	0.00	N.S.	N.S.
Polygonum douglassii	0.00	0.00	0.00	0.00	0.47	1.84	1.02	2.41	N.S.	N.S.
Rumex paucifolius	0.25	0.38	0.49	0.74	0.75	2.91	1.33	3.15	N.S.	N.S.
Rorippa curvisiliqua	0.00	0.00	0.16	0.23	0.79	3.06	0.00	0.00	N.S.	N.S.
Senecio integerrimus	0.18	0.27	0.25	0.38	0.00	0.00	0.00	0.00	N.S.	N.S.
Achillea millefolium	0.88	1.32	0.16	0.23	0.00	0.00	0.61	1.44	N.S.	N.S.
Taraxacum officinale	0.00	0.00	0.31	0.47	0.00	0.00	0.22	0.52	N.S.	N.S.
Ranunculus cymbalaria	0.00	0.00	0.00	0.00	7.24	28.08	5.07	12.02	N.S.	N.S.
Mimulus shevockii	0.16	0.23	0.27	0.40	0.00	0.00	0.42	0.99	N.S.	N.S.
Mimulus mephiticus	0.00	0.00	0.09	0.13	0.35	1.37	0.11	0.26	N.S.	N.S.
Phlox gracilis	0.00	0.00	0.00	0.00	0.00	0.00	3.06	7.25	N.S.	N.S.

Table 9. Total vegetation cover and species richness per 0.25 square meter quadrat in grazed and exclosed sections of Templeton and Mulkey Meadows, 1993.

MEADOW	TREATMENT	FOLIAR COVER	RICHNESS/ QUADRAT	TOTAL RICHNESS
MULKEY	PROTECTED	66.46	5.15	53
	GRAZED	66.75	5.39	60
TEMPLETON	PROTECTED	25.80	3.61	49
	GRAZED	42.14	4.04	52

Table 10. Mean standardized importance values and congruence with responses predicted by grazing response class for the top 20 species in grazed and ungrazed reaches, Templeton and Mulkey Meadows

SPECIES	MEAN GRAZED	MEAN UNGRAZED	OVERALL MEAN	EXPECTED RESPONSE	OBSERVED RESPONSE
Artemisia cana	32.17	28.03	30.10	D	D
Muhlenbergia richardsonis	23.72	20.55	22.14	D	D
Carex simulata	14.47	19.28	16.87	I	I
Trifolium longipes	14.64	12.18	13.41	D	D
Deschampsia Caespitosa	9.66	12.44	11.05	I	I
Gayophytum decipiens	11.52	9.96	10.74	D	D
Trifolium monanthum	9.86	8.03	8.95	D	D
Arnica mollis	13.07	4.04	8.55	D	D
Carex utriculata	6.70	9.82	8.26	I	I
Potentilla anserina	8.39	7.52	7.96	D	D
Torreyochloa pallida	6.16	8.67	7.41	I	I
Arnica sororia	7.14	6.33	6.73	D	D
Agrostis idahoensis	6.27	5.70	5.99	D	D
Erigeron peregrinus	6.98	4.82	5.90	D	D
Carex athrostachya	4.95	5.94	5.44	I	I
Ivesia campestris	5.02	4.44	4.73	D	D
Poa pratensis	5.79	3.53	4.66	D	D
Aster occidentalis	5.25	3.71	4.48	D	D
Happlopappus apargioides	5.81	1.55	3.68	D	D
Hordeum brachyantherum	5.11	1.88	3.49	D	D
KEY			PERCENT CORRECT		100%
D = DECREASE					
I = INCREASE					

Table 11. Willow growth and survival inside and outside experimental exclosures. Survival estimates assume 1000 cuttings planted per treatment, 500 in 1982 by Forest Service resource crews and 500 in 1984 by our crew; n = 2 for each treatment in 1985, representing north and south side of stream, but both sides counted together in 1993.

	RAMSHAW MEADOW		TEMPLETON MEADOW	
	Inside	Outside	Inside	Outside
1985				
Survival (% ± 1 sd)	45 + 6	22.5 + 3	17.1 ± 1.3	13.3 ± 0.3
Location [1]				
Edge:Bench:Bank	34:55:11	9:25:17	12:42:40	8:38:9
1993				
Survival (%)[2]			10.9	2.0
Plant volume (M^3. ± 1 se)[3]			0.99 ± 0.09	0.63 ± 0.2
Plant height (M. ± 1 se)[3]			0.76 ± 0.02	0.56 ± 0.06
Location - Edge:Bench:Outer Bank (Ratio)			6:57:37	10:80:10

[1] % survival in microhabitats estimated as no. live/no. live + dead cuttings found in each zone;

[2] calculated from plants with 10-year old tags or cut stem still evident, but survival was 16.2% vs. 3.2 % if questionable plants were included.

[3] plant dimensions based on product of mean height (3 measurements in the center of the shrub) and the lateral and longitudinal dimensions (perpendicular x parallel to stream)

Table 12. Mean densities (± 1 SE) of selected common invertebrate taxa in grazed vs. exclosure sections of experimental meadows.
Other taxa present but no in sufficient numbers for analysis included: Ameletus, Epeorus, Serratella, Ephemerella, Cinygma, Epeorus, Malenka, Chironominae, Ophiogomphus, Hydraena, Hydropsyche, Dicosmoecus, Rhyacophila, and numerous micro-crustaceans and terrestrial insects.

Taxon	Ramshaw Mdw Grazed	Ramshaw Mdw No Grazing	Templeton Mdw Grazed	Templeton Mdw No Grazing	Treatment F - p	Site (Meadow) F - p
Ephemeroptera *Baetis*	58.7 (18.6)	87.3 (29.8)	21.0 (7.5)	28.0 (12.0)	1.27, 0.3	**5.64, 0.03**
Paraleptophlebia	25.2 (13.3)	10.5 (5.6)	1.5 (0.5)	1.7 (0.7)	0.6 , 0.4	2.80, 0.11
Drunella	2.8 (1.8)	2.8 (2.2)	3.2 (1.8)	7.3 (3.8)	0.7 , 0.4	0.04, 0.8
Plecoptera *Sweltza*	32.6 (12.7)	14.8 (5.8)	1.2 (0.5)	2.0 (1.0)	1.01, 0.3	**4.94, 0.04**
Coleoptera *Heterlimnius*	168.9 (45.7)	287.8 (99.2)	37.5 (13.2)	62.3 (19.8)	2.21, 0.15	**7.10, 0.01**
Diptera *Simulium*	13.6 (4.9)	10.5 (4.3)	1.0 (0.5)	2.7 (0.4)	0.1 , 0.7	**5.43, 0.03**
Bezzia	10.9 (3.4)	7.3 (1.9)	10.0 (7.2)	4.3 (2.3)	0.9 , 0.3	0.17, 0.7
Tanypodinae	14.4 (5.7)	17.3 (1.9)	10.0 (3.5)	16 (9.0)	0.04, 0.8	0.18, 0.7
Orthocladiinae	192 (48.0)	130.3 (34.5)	27.7 (9.2)	44.7 (19.1)	0.8 , 0.4	2.91, 0.12
Hexatoma	3.1 (1.3)	2.5 (1.0)	7.8 (2.0)	9.0 (1.0)	0.1 , 0.9	**11.1, 0.01**
Empididae	4.3 (1.6)	1.8 (0.5)	6.8 (3.4)	14.4 (4.6)	0.7 , 0.4	**8.73, 0.01**
Psychodidae	9.0 (5.8)	11.5 (5.8)	0	0.7 (0.5)	0.1 , 0.7	2.88, 0.10
Ostracoda	16.5 (7.7)	6.3 (3.2)	4.0 (2.0)	2.7 (0.3)	1.57, 0.2	1.69, 0.2
Acari	92.7 (39.0)	93.7 (44.0)	30.7 (11.6)	46.3 (20.6)	0.02, 0.9	2.03, 0.17
Pisidium	2.7 (1.2)	1.0 (0.3)	8.2 (5.4)	3.3 (1.2)	0.9 , 0.4	1.44, 0.2
Oligochaeta	10.7 (4.0)	5.8 (2.1)	1.8 (0.9)	4.0 (1.0)	0.4 , 0.5	3.10, 0.09

Table 13. Mean number of invertebrates per 0.2 m2 on some major substrate types or 'patches', and area occupied by each patch type (± 1 S.D.) in presence vs. absence of grazing in 3 major meadows.

Substrate	Ephemeroptera	Chironomidae	Simuliidae	Taxon Richness	% Area - No Grazing	% Area - Grazed
Moss	1110	1081	112	32	1.2 (1.6)	1.6 (2.9)
Ranunculus	564	2020	137	30	3.4 (2.0)	6.2 (2.7)
Carex	160	855	33	31	8.3 (5.4)	6.2 (3.7)
Sand	28	660	4	18	74.3 (10.9)	74.0 (8.4)
Cobble	456	1130	15	27	6.8 (10.7)	6.0 (7.8)

Table 14. Drifting patterns of some common invertebrates in Mulkey Meadow. Values are drift density, or mean number of individuals per m^3, during each period.

Taxon	Morning (9-11 AM)	Day (1-3 PM)	Dusk (8-9 PM)	Night (12-2 AM)
Baetis	0.04	0.07	0.48	0.58
Paraleptophlebia	0.03	0.05	0.32	0.64
Chironomidae	0.09	0.17	1.56	0.94
Simulium	0.06	0.04	0.08	0.05
Heterlimnius	0.04	0.03	0.09	0.06
Ostracoda	0.67	0.88	1.22	1.76
Micro-Crustacea	0.28	0.17	0.31	0.10
Acari	0.12	0.16	0.65	0.50
Aquatic insects	1.34	1.57	4.70	4.74
Terrestrial insects	0.06	0.15	0.21	0.06

Table 15: Summary table of exclusion effects in the three study meadows. (N.S.=a notable but nonsignificant difference, N/A=not tested)

MEADOW	Mulkey	Templeton	Ramshaw
VARIABLE			
Soils			
soil-N	N/A	no difference	N/A
soil-OM	N/A	no difference	N/A
N-mineralization	N/A	no difference	N/A
C/N ratio	N/A	no difference	N/A
soil-pH	N/A	no difference	N/A
soil infiltration-general	N/A	no difference	N/A
soil infiltration-cattle trails	N/A	increase	N/A
Geomorphology			
channel aggradation	no difference	no difference	N/A
lateral erosion	N/A	no difference	N/A
channel depth	increase	N/A	increase
bank water depth	increase	N/A	increase
canopy shading	increase	N/A	increase
stream width:depth	decrease	N/A	decrease
channel width:depth	decrease (n.s.)	N/A	decrease (n.s.)
bank angle	decrease (n.s.)	decrease (n.s.)	decrease (n.s.)
bank overhang	decrease (n.s.)	N/A	decrease (n.s.)
geomorphology index	increase	N/A	increase
Vegetation			
pioneer vegetation	N/A	increase	increase
vegetation cover	N.S.	N.S.	N/A
hydric graminoid vegetation	increase	increase	N/A
mesic forb vegetation	decrease	decrease	N/A
willow herbivory	N/A	decrease	decrease
willow recruitment	increase	N/A	increase
willow cutting survival	N/A	increase	increase
Aquatic Fauna			
invertebrate drift densities	no difference	no difference	no difference
invertebrate benthic densities	no difference	no difference	no difference
trout densities	increase	N/A	increase
trout biomass	increase	N/A	increase